VOLUME I

BASELINE
ECOLOGICAL RISK
ASSESSMENT

SAUGET AREA 2
SITES

SAUGET, IL

REPORT
APPENDICES

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DRAFT BASELINE
ECOLOGICAL RISK ASSESSMENT

SAUGET AREA 2 SITES
(SITES O, P, Q, R, AND S)
SAUGET, ILLINOIS

VOLUME I

REPORT

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EXECUTIVE SUMMARY

A Baseline Ecological Risk Assessment (BERA) was prepared for the Sauget Area 2 Sites (the Sites) located in the villages of Sauget and Cahokia, Illinois. Environmental concerns at the Sites are being investigated subject to an Administrative Order on Consent (Docket No. V-W-'01-C-622) between the Sauget Area 2 Sites Group (the Group) and the U. S. Environmental Protection Agency (USEPA) Region V, pursuant to Sections 106 and 122 of the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA).

The objective of the BERA is to evaluate the potential for adverse ecological effects to biological receptors living within the aquatic and terrestrial ecosystems located on or adjacent to the Sites, as a result of exposures to Site-related constituents. The BERA is a baseline evaluation of ecological risks that utilizes both historical data regarding the Sites and data that were collected as part of investigative activities within the Mississippi River and the five Sites. The BERA was prepared using conservative, but realistic, assumptions about potential exposures and assumed that no remedial action has occurred.

This BERA was completed in accordance with a USEPA-approved Ecological Risk Assessment Work Plan, which was included as Section 12.0 to the Support Sampling Plan (SSP) (URS, 2002). Data used in the completion of this BERA included laboratory analytical data that described the concentrations of contaminants of potential ecological concern (COPECs) found within various abiotic and biotic matrices associated with the Sites and the Mississippi River.

AQUATIC ECOLOGICAL RISKS IN THE MISSISSIPPI RIVER

Potential ecological risks to aquatic receptors within the Mississippi River were assessed through the collection of surface water and sediment samples from locations upstream, adjacent to, and downstream of the five disposal Sites. The samples were chemically analyzed to determine the concentrations of COPECs possibly present. Bioassays were run on both surface water and sediment samples to evaluate acute and chronic toxic effects to the endpoint species. Additionally, bioaccumulation tests were conducted to determine the body burdens of COPECs in test organisms exposed to sediments for an extended period of time. Fish tissue body burdens identified in historic sampling activities was also evaluated to assess potential ecological impacts.

An Interim Groundwater Remedy is currently being implemented downgradient of Sauget Area 2 Sites O, Q (North), R and S to control adverse impacts on the Mississippi River due to groundwater discharges from these Sites; Sauget Area 1 Sites G, H, I and L; and industrial facilities in Sauget and Cahokia, Illinois.

The results of the BERA for the different evaluated media are presented below.

Sediments – The BERA concluded that there were no adverse ecological impacts associated with the presence of COPECs in sediments.

Chemical analysis of sediments indicated that there were measurable concentrations of COPECs that exceeded conservative ecologically based benchmarks. The COPECs included volatile organic compounds (VOCs) such as acetone and chlorobenzene; semivolatile organic compounds (SVOCs) such as 1,2-dichlorobenzene; pesticides such as dieldrin, endrin aldehyde, heptachlor epoxide; herbicides such as MCPP; and metals such as arsenic, barium, cadmium, copper, lead, manganese, nickel, and zinc. The highest detected concentrations of organic COPECs were located along transects closest to the shore in the sampling area located downgradient of Site Q (North) and just downstream of Site R. None of the inorganic COPECs exceeded their respective benchmarks by a significant degree and the pattern of distribution throughout the sampling plots adjacent to or downstream of the Sauget Area 2 Sites appeared to be random.

However, the sediment bioassays (considered to be a stronger indicator of potential toxic effects) demonstrated that there were no significant toxic effects in any of the Site-related sediment samples. For the acute toxicity test, there were no significant differences in mean survival when Site-related samples were compared to their respective control samples for any of the sampling sites adjacent to, or downstream of, the disposal areas. Similarly, the chronic test concluded that none of the sediment samples collected from any of the sampling plots exhibited mean growth that was significantly lower than the mean growth of the corresponding laboratory control samples.

Surface Water - The BERA concluded that there were limited ecological impacts associated with the presence of COPECs in surface water.

Surface water COPECs identified through chemical analyses included p-chloroaniline, 2,4-D, aluminum (total), barium (dissolved, total), copper (total), iron (total), manganese (total), and vanadium (dissolved, total). P-chloroaniline had the greatest exceedance of its conservative screening benchmark, followed closely by 2,4-D. Maximum concentrations of these two constituents were detected at the sampling area downgradient of Site Q (North) and just downstream of Site R on the transects closest to the riverbank. Barium had the greatest exceedance of its benchmark, while the remaining metals only slightly exceeded their respective benchmarks.

Surface water bioassays indicated that acute toxicity was limited to the sampling area downgradient of Site Q (North) and just downstream of Site R. The sample with the lowest survival and young production corresponded to the surface water sample that had the highest concentrations (by nearly an order of magnitude) of p-chloroaniline and 2,4-D. Chronic toxicity was also seen at other sampling locations downstream where detected concentrations of p-chloroaniline and 2,4-D were noted.

Conclusion of the Aquatic Risk Assessment - The BERA concluded that no adverse ecological impacts were identified with sediments within the Mississippi River and only limited surface water impacts were identified. Two organic compounds (p-chloroaniline and 2,4-D) were identified as the principal constituents of concern in the surface water environment of the Mississippi River adjacent to the Sauget Area 2 Sites.

Historical sampling performed at Sauget Area 2 Site R, which is immediately upstream of Sampling Area R3, indicates that p-chloroaniline and 2,4-D are present at this site. Sediment and surface water sampling performed by Menzie-Cura in October and November 2000 indicated that groundwater discharging to surface water downgradient of Site R resulted in an adverse impact on the Mississippi River. Based on this information, USEPA issued a Unilateral Administrative Order (Docket No. V-W-'02-C-716) on September 30, 2002 for performance of an Interim Groundwater Remedy, consisting of installation of a physical barrier and groundwater extraction system downgradient of Site R, to protect the Mississippi River. Groundwater extraction started on July 15, 2003, and construction of the physical barrier is scheduled to start on September 2, 2003 and be completed in the first quarter of 2004. The implementation of the interim groundwater remedy will eliminate the discharge of contaminated groundwater into the river. This will eliminate the potential ecological risks identified with these two compounds. For that reason, no additional remedial action is considered necessary to protect the aquatic ecosystem in the Mississippi River.

ECOLOGICAL RISKS IN THE FLOODPLAIN

The BERA evaluated the potential for COPECs to impact Receptors of Interest (ROIs) with small home ranges (prairie vole and short-tailed shrew) and large home ranges (osprey, mink and red fox). Potential for adverse impacts was evaluated on a site-by-site basis for the vole and shrew because of their small foraging areas and on study area basis for the osprey, mink and fox because of their large foraging areas. For the small-ranging organisms at the individual sites, the prairie vole was considered the most appropriate indicator of potential ecological risks because habitat suitable to support the short-tailed shrew was not dominant at the five disposal areas (Sites O, P, Q, R and S). Risks to these organisms were calculated based on food chain models using concentrations of COPECs identified in surface soil, plant tissues, and invertebrate body burdens as input parameters.

Potential floodplain ecological risks are summarized below.

Piscivores - A limited number of COPECs were identified for consumption of fish and surface water by the mink and osprey, two organisms that were evaluated based on aquatic exposures. From a habitat standpoint, the riverbank adjacent to the Sauget Area 2 Sites is not good habitat for any fish-eating mammal. Much of the 14,000 linear feet of riverbank is covered with stone riprap, removing cover requirements that this animal has. The remainder of the bank contains piers, pilings, buildings and other human disturbances, which would further preclude fish-eating mammals from inhabiting the area.

Nitrobenzene, MCPP, PCBs, dioxin/furans, aluminum and antimony were all identified as COPECs for the mink. However, most of the estimated ecological risks for the mink were based on consumption of fish from the large pond. The large pond is one of two ponds located in the southern end of Site Q. Identified as Site Q (Ponds), these ponds are ephemeral water bodies that will support a fish community on a temporary basis only if fish are washed into the ponds through overbank flooding of the Mississippi River. Fish were collected from the large pond, prior to it's drying up, and analyzed for the presence of COPECs. If those fish are removed from the modeling, as the community no longer exists, then the only COPECs identified for the mink are MCPP and antimony. The adverse risks noted with those constituents were slight.

For the osprey, mercury was the only COPEC. The potential for an ecological risk was small. Since surface water concentrations and bioaccumulation factors were used to calculate fish tissue mercury concentrations, actual risks due to mercury are likely to be lower than the predicted risks.

Plants - The potential for direct impact to plants was evaluated by comparing surface soil concentrations to screening plant benchmarks. A variety of COPECs in each disposal site were identified with concentrations in excess of these benchmarks. Site S had the highest number of organic COPECs that exceeded the plant benchmarks, while Site Q had the highest number of inorganic COPECs in excess of the conservative screening plant benchmarks.

These benchmarks are considered to be highly conservative even by the authors of the benchmarks. While a number of COPECs were identified, no indication of impacts to plants was noted in field observations conducted at the Sites. The vegetative communities in each of the disposal areas were marked by robust and vigorous plant growth with no indications of phytotoxic effects. The prairie vole food chain model provides a more accurate assessment of potential plant impacts by evaluating the presence of COPECs that were identified in plant tissues as they relate to a higher trophic level receptor.

Herbivores - In examining the potential for ecological risks at the five disposal sites, only limited risks were identified at Site P or Site Q (South) for the prairie vole. Potential ecological risks were predicted at Site O (PCBs, dioxin/furans, mercury, and thallium) and Site S (pentachlorophenol, PCBs, and mercury). At Site O, only PCBs and dioxins/furans exceeded both the NOAEL and the LOAEL benchmark values for the prairie vole. Potential areas of ecological risk at Site O are centered on sampling locations W-O-1 and W-O-3 and are shown on Figures ES-1 through ES-3. Adverse risks were also predicted for Site R (cobalt and mercury), however Site R is covered with a dirt cap. Further, the cap is regularly mowed and, consequently, is not considered a viable habitat for the vole. The potential adverse risks estimated at Site R were not considered to be significant.

Carnivores - An assessment of the potential for site-wide adverse ecological impacts to the red fox were conducted to determine whether cumulative affects from the five disposal Sites would be noted. The assessment was made based on modeled exposure to prey items (the short-tailed shrew and the prairie vole). In keeping with OSWER Directive 9285.7-28P, as an upper trophic level organism, the red fox was considered be the more critical receptor while the importance of the two small mammals was as prey items.

Aluminum had the highest exceedance of both its ecotoxicity benchmark values. PCBs and dioxin/furans, which were expected to be in prey tissue based on the model parameters, also exceeded their ecotoxicity benchmarks. Site O and Site S were the only sites where PCBs were modeled to be present at elevated concentrations (in excess of TRV benchmarks) in both the shrew and the vole and these two sites served as the greatest contributor of PCB and dioxin/furan risks to the red fox. Since the risks for PCBs were predicted based on the shrew and the vole as a prey base for the fox, areas potentially needing remedial action to protect these organisms from PCBs and dioxins/furans would also potentially protect the red fox. These areas are shown on Figures 9-2 and 9-3.

It is noted that the red fox has a mean home range of 1,727 acres; it is highly unlikely that the disposal sites (total area less than 150 acres) would support a large population of red fox. Noting the discontinuity of the sites, it is more likely that a small number of fox utilize a portion of different disposal areas for foraging, moving between contaminated and non-contaminated areas. Additionally, the fence surrounding Site R would limit access of the fox to this disposal area.

Ponds - The BERA also evaluated potential ecological risks to aquatic receptors associated with the aforementioned ponds. While sediment and surface water screening against conservative benchmarks indicated the presence of some organic and inorganic COPECs, acute and chronic toxicity testing of both matrices did not indicate any adverse effects.

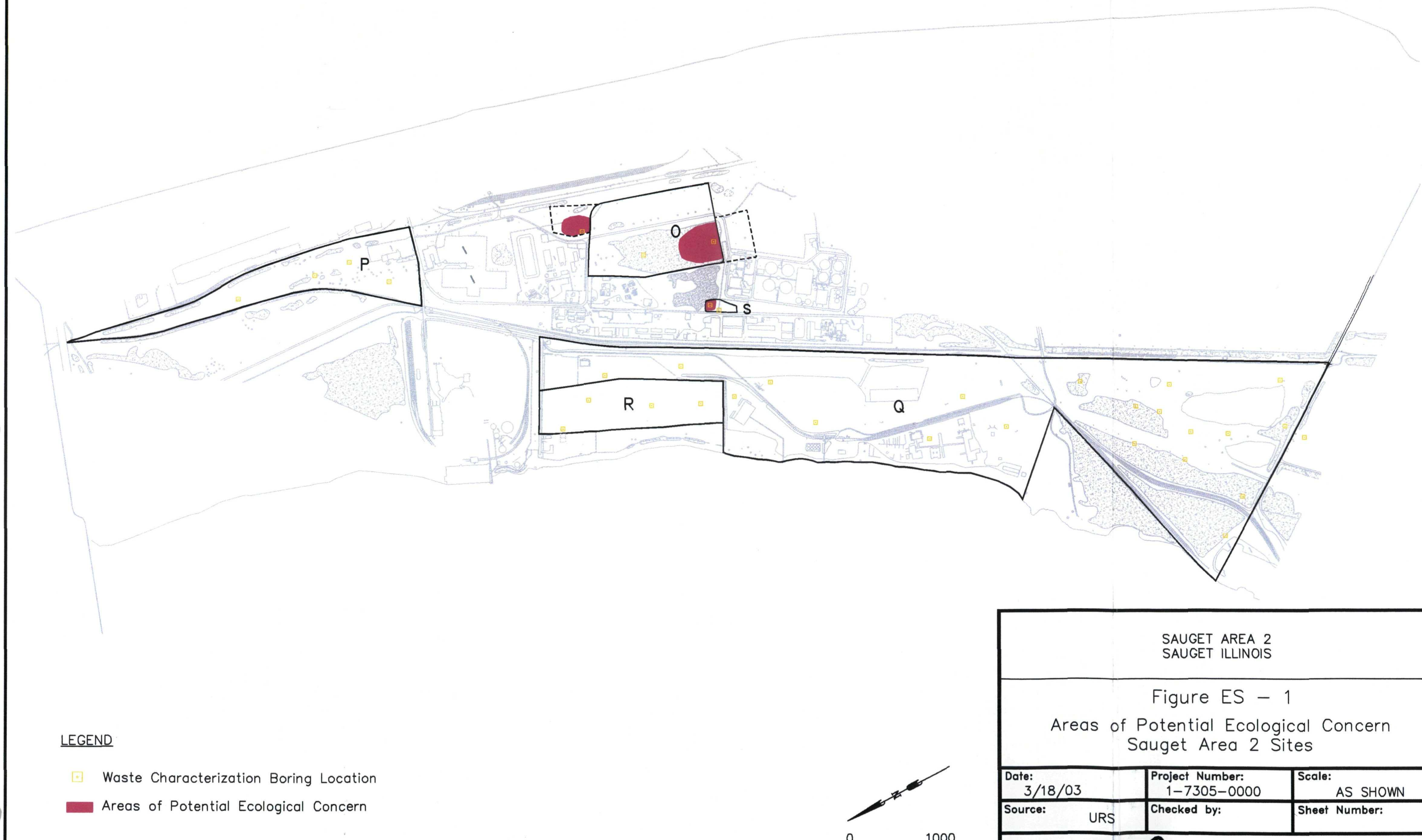
However, as the ponds were mostly dried by the time this BERA was implemented and only a partial data set could be collected to evaluate them. Surface water and sediment quality data were collected in June 2003. These data will be presented in an addendum to this BERA at a future date.

Conclusion of the Floodplain Risk Assessment - The BERA concluded that potentially significant ecological impacts were identified for Site O and for Site S. This determination was based on food chain modeling to the prairie vole and to the red fox.

CONCLUSIONS OF THE BERA

The BERA concluded that no adverse ecological impacts were associated with sediments within the Mississippi River. Limited surface water impacts based on toxicity testing were identified with p-chloroaniline and 2,4-D identified as the principal constituents of concern in surface water. With the implementation of the interim groundwater remedy at Site R, no additional remedial actions are considered necessary to protect the aquatic ecosystem in the Mississippi River.

The BERA also identified the potential for adverse ecological impacts associated with the presence of COPECs in surface soil found in Site O and Site S. For Site O, the most significant COPECs included dieldrin, lindane, PCBs, dioxins/furans, aluminum, and mercury. For Site S, the most significant COPECs included pentachlorophenol, beta-BHC, endrin, and lindane, and PCBs. These areas will be evaluated further in the Feasibility Study for the identification of potential remedial actions. Limited ecological risks were identified with surface water and sediments in Site Q (Ponds), however, a further determination of potential ecological risk will be made upon the evaluation of surface water and sediment quality data collected in June 2003.



LEGEND

Waste Characterization Boring Location

Areas of Potential Ecological Concern

SAUGET AREA 2
SAUGET ILLINOIS

Figure ES – 1

Areas of Potential Ecological Concern
Sauget Area 2 Sites

Date: 3/18/03	Project Number: 1–7305–0000	Scale: AS SHOWN
Source: URS	Checked by:	Sheet Number:

amec EARTH & ENVIRONMENTAL, INC.
285 Davidson Avenue, Suite 100
Somerset, NJ 08873

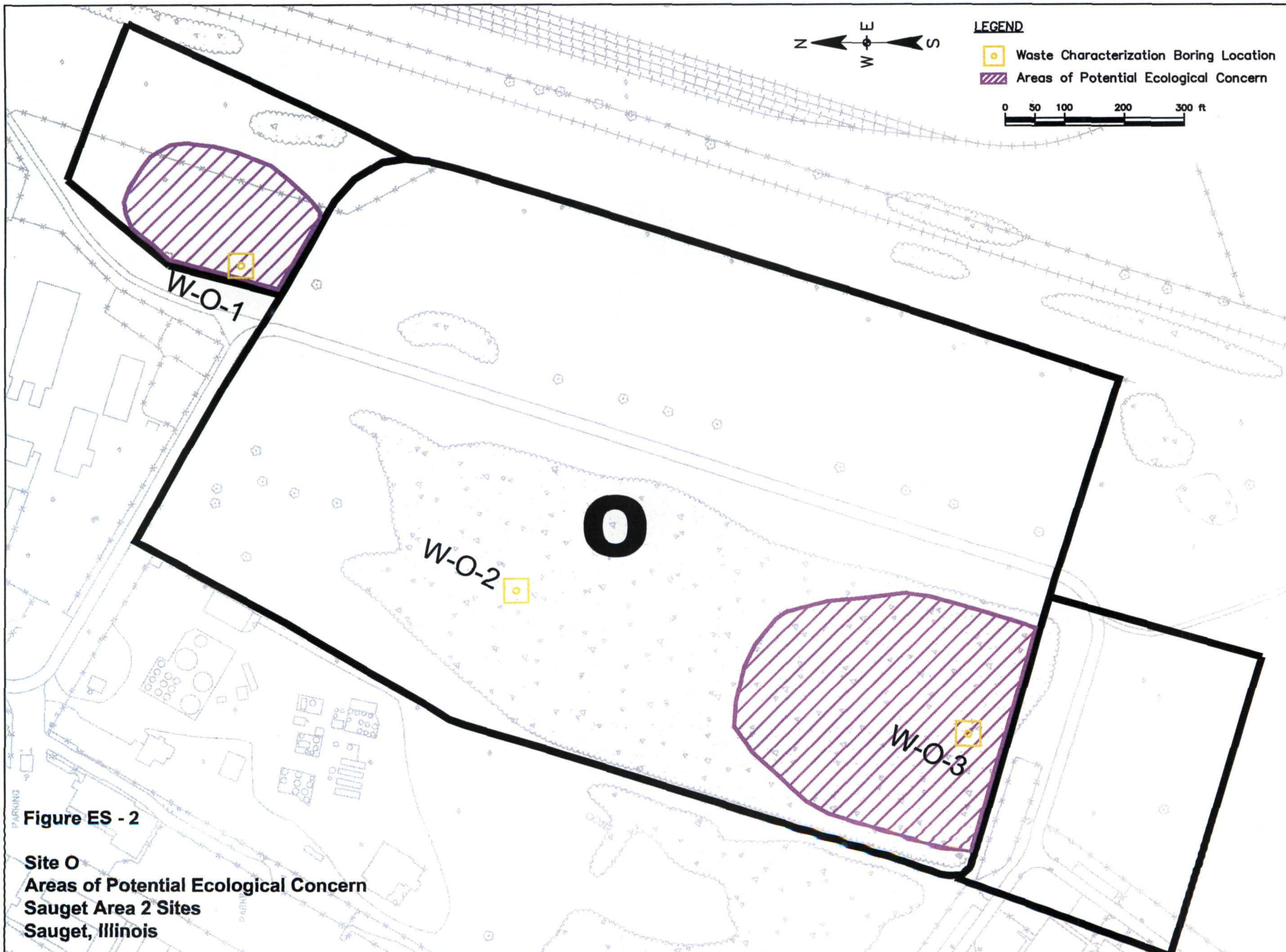


Figure ES - 2

Site O
Areas of Potential Ecological Concern
Sauget Area 2 Sites
Sauget, Illinois

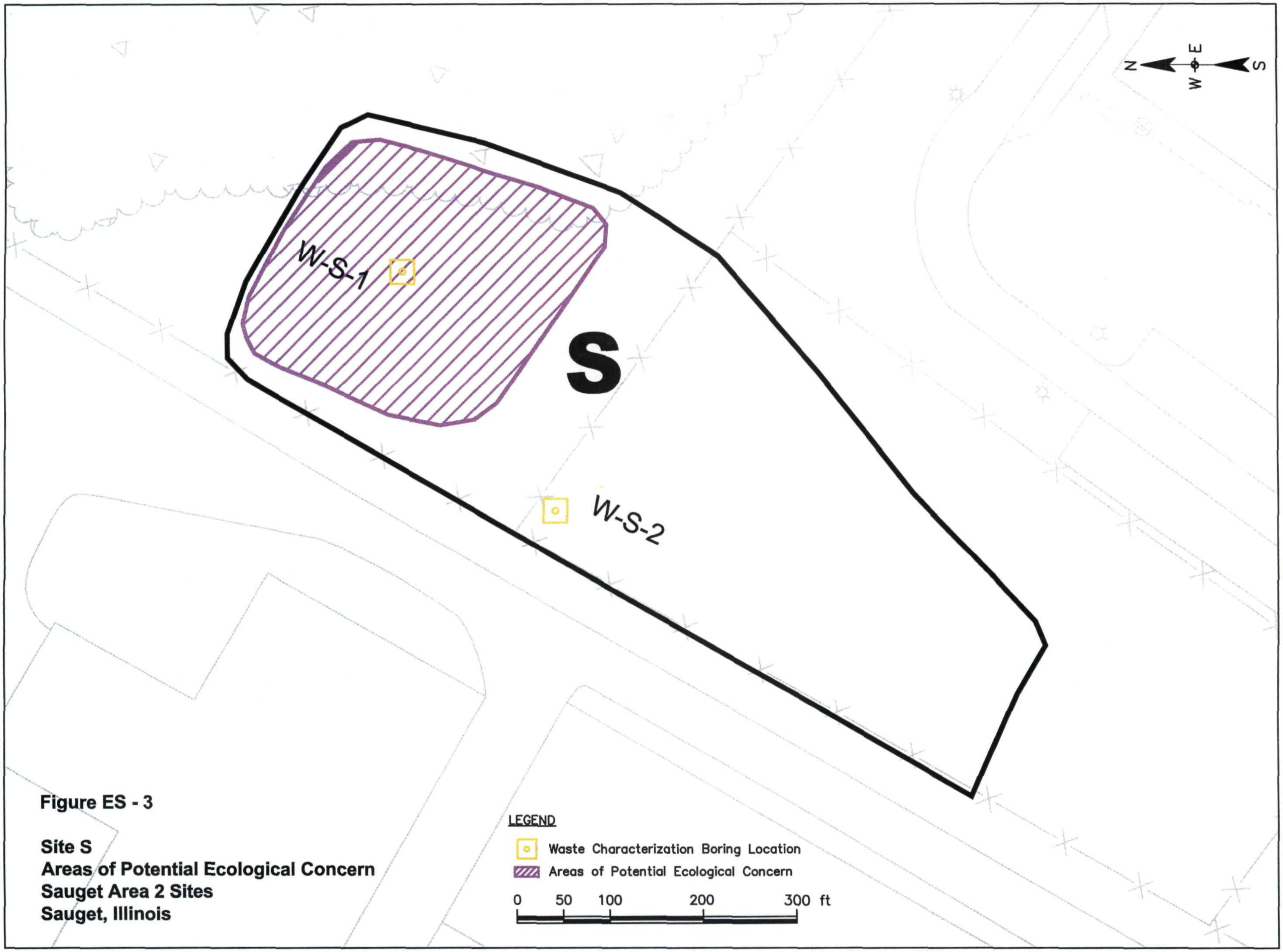


Figure ES - 3

Site S
Areas of Potential Ecological Concern
Sauget Area 2 Sites
Sauget, Illinois

Section 1

1.0 INTRODUCTION

A Baseline Ecological Risk Assessment (BERA) was prepared for the Sauget Area 2 Sites (the Sites) located in the villages of Sauget and Cahokia, Illinois. Environmental concerns at the Sites are being investigated subject to an Administrative Order on Consent between the Sauget Area 2 Sites Group (the Group) and the U. S. Environmental Protection Agency (USEPA) Region V, pursuant to Section 106 of the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). The Sites include five identified terrestrial source areas (Sites O, P, Q, R, and S). Two of the disposal sites have been divided to include smaller subsections. Site O has been subdivided to include Site O and Site O (North). Site Q has been subdivided into Q (North), Q (Central), Q (South), and Q (Ponds). Based on the fact that Q (North) and Q (Central) are disturbed (covered with pavement, roads, parking areas, debris and buildings) and contained no viable habitat for wildlife, the USEPA agreed that this BERA would only assess Q (South), and Q (Ponds). Since the disposal sites front approximately 14,000 linear feet of the Mississippi River, the BERA included an aquatic assessment. Six sampling areas or plots, one upstream of the Sites (labeled R1) and five adjoining the Sites (labeled R2, R3, R4, R5, and R6) were evaluated in the Mississippi River.

1.1 PURPOSE AND OBJECTIVES

The objective of the BERA is to evaluate the potential for adverse ecological effects to occur as a result of exposures to Site-related constituents by biological receptors living within the aquatic and terrestrial ecosystems located on, or adjacent to, the Sites. The BERA utilized both historic data regarding the Sites and data that were collected as part of investigative activities within the Mississippi River and the five Sites through November 2002. This BERA was prepared using conservative, but realistic, assumptions about potential exposures and assumed that no remedial action has occurred.

As outlined in the USEPA-approved Ecological Risk Assessment Work Plan (Appendix I-A), the principal functions of the BERA are to:

- Determine whether actual or potential ecological risks currently exist at the Sites;

- Identify those constituents present at the Sites that pose potential ecological risks; and
- Generate data and information for risk management and risk reduction decisions.

1.2 Conceptual Approach to Ecological Risk Assessment

This BERA was completed in accordance with a USEPA-approved Ecological Risk Assessment Work Plan, which was included as Section 12.0 to the Support Sampling Plan (SSP) (URS Corporation, 2002). Data used in the completion of this BERA included laboratory analytical data that described the concentrations of compounds of potential ecological concern (COPECs) from various abiotic and biotic matrices associated with the Sites and the Mississippi River.

Field data for this BERA were collected in accordance with the SSP and included:

- Qualitative biological surveys of flora and fauna on the Sites;
- Surface water and sediment samples collected for laboratory analysis and bioassay analysis from the Mississippi River and ponds located in the southern end of Site Q;
- Sediment samples collected for bioaccumulation studies from the Mississippi River and ponds located in the southern end of Site Q;
- Fish tissue collected from the ponds located in the southern end of Site Q;
- Surface soil samples collected from the six disposal sites;
- Plant tissue samples collected at locations concurrent with the surface soil samples;
- Terrestrial invertebrate tissue samples collected from the six disposal sites; and
- Surface soil samples collected from the six disposal sites that were used to conduct earthworm bioaccumulation studies.

Historic data utilized for in this BERA include fish tissue body burden data developed by Menzie-Cura as part of the Sauget Area 1 Ecological Risk Assessment (Menzie-Cura, 2001).

Further information regarding the collection of this data can be found in *Floodplain Field Sampling Report* (AMEC, 2003a) and *Aquatic Field Sampling Report* (AMEC, 2003b).

This BERA was structured utilizing the standard paradigm for ecological risk assessment as outlined in *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA, 1997a). The BERA consists of the following general elements:

- Problem formulation;
- Ecological exposure assessment;
- Ecological effects assessment; and
- Risk characterization.

The BERA begins with a problem formulation phase (Section 2.0) that characterizes the environmental setting of the disposal sites, as well as the Mississippi River. The problem formulation phase develops a conceptual site model that addresses the contaminants known to exist at the Site and identifies contaminant fate and transport mechanisms, mechanisms of toxicity, and complete exposure pathways. As a final part of the problem formulation phase, endpoints for assessing ecological attributes in this BERA were selected.

Section 3.0 and Section 4.0 discuss the COPECs that have been identified in the aquatic environment of the Mississippi River and the terrestrial and aquatic environment of the floodplain that contains the Area 2 Sites. The effects assessment (Section 5.0 and Section 6.0) characterizes the relationship between chemicals identified during the investigative sampling and various attributes associated with ecological receptors. The ecological effects evaluation identifies those exposure levels that represent conservative thresholds for adverse ecological effects.

Risk characterization (Section 7.0) is the phase of risk assessment in which the information concerning exposure and the information concerning potential effects of exposure are integrated to estimate risks (the likelihood of effects given the exposure) or potential impacts.

The final sections of the BERA include an uncertainty section (Section 8.0), which summarizes the uncertainties noted in the development of the aquatic and floodplain ecological risk estimates. Section 9.0 presents the conclusions of the BERA. The BERA has identified that the primary risk drivers for the Mississippi River include p-chloroaniline and 2,4-D associated with a groundwater discharge from Site R and for the floodplain area, polychlorinated biphenyls (PCBs) and dioxins/furans.

Section 2

2.0 PROBLEM FORMULATION

The problem formulation section consists of the description of the relevant features and current condition of the environment, the description of the potential sources for Site-related COPECs, the identification of ecological receptors at the Site and surrounding area and the development of the Conceptual Site Model (CSM).

2.1 SITE SETTING

Sauget Area 2 is situated adjacent to the Mississippi River within the villages of Sauget and Cahokia, Illinois. The boundaries of Area 2 are generally located east of the Mississippi River, south of the MacArthur bridge railroad tracks, west of Illinois State Highway 3, and north of Cargill Road. Area 2 fronts over 14,000 linear feet of the eastern bank of the Mississippi River.

Sauget Area 2 includes five disposal areas, Sites O, P, Q, R, and S (the "Area 2 Sites"), which are adjacent to, or in close proximity of, the Mississippi River. Figure 2-1 is a Site drawing and depicts the locations of the various disposal sites. Two of the Sites, Site Q and Site R, are located on the west side of the floodwall and levee that is operated and maintained by the US Army Corps of Engineers and the Metro East Sanitary District. The floodwall is designed to protect the City of East St. Louis and the Villages of Sauget and Cahokia from flooding. The other disposal sites (O, P, and S) are located on the dry side of the floodwall and levee.

The United States Geological Survey (USGS, 1999) notes that the characteristics of sediments and surface water in the Mississippi River below its confluence with the Missouri River have long differed from the reach upstream of the confluence. Those differences are the result of influences of the City of St. Louis and the Missouri River drainage basin. The City of St. Louis has had a significant effect on river water quality in this segment of the river due to sewage and industrial discharges from within the city. Missouri Department of Natural Resources (1994) estimates that 300 tons of ground garbage was discharged in the river daily in 1957, and as late as 1970, raw sewage was discharged directly into the river by the City of St. Louis (Corbett, 1997). The Missouri River drains an area with highly erodible soils and is the major source of sediments to the Mississippi River. This contribution of sediment leads to changes in water clarity, sedimentation of shallow areas, and the introduction of non-Site-related sediment-borne constituents.

Boyer (1984) reported that the City of St. Louis contributes significant amounts of constituents from wastewater effluents, industrial discharges and urban runoff, including metals and organic compounds such as PCBs, to the Mississippi River. Pesticides and herbicides have been found to be significant contaminants in the Mississippi River. The reach upstream of the Missouri confluence has been reported to contribute 40 to 50 percent of the pesticide and herbicide load within the Mississippi River, even though it represents only 22 percent of the flow from the entire river (Goolsby and Pereira, 1995). USGS (1999) notes that surface water quality in the Mississippi River have improved since the passage of the Federal Water Pollution Control Act in 1970, though concentrations of pesticides still exceed USEPA guidelines during low flow, high use periods of the year.

Field observations indicate that the disposal sites have been significantly impacted by anthropogenic activities. These activities included clearing and construction of roads and railroad lines, construction of buildings, and the development of industrial activities. Most of the disposal sites show signs of extensive clearing and/or disturbance, and, with the exception of Q (South) and Q (Ponds), they are vegetated either solely by herbaceous communities or by herbaceous communities with a thin layer of early successional shrubs or trees. Approximately 50% of Q (South) is covered with a mature secondary hardwood forest. An herbaceous community covers the remaining section of Q (South). The ponds located at the southern end of Q (South) (identified as Q (Ponds)) are ephemeral bodies of water that are subject to cycles of filling and drying, with the length of the cycle being dependent upon rainfall and the flood stage of the Mississippi River. When the ponds are filled with water, they can provide some habitat for amphibians and fish (when washed in by floods). During these periods the ponds also provide a water source for wildlife. When the ponds are dry, the land surface that was flooded is covered with herbaceous vegetation.

Table 2-1 lists all vegetative species that were observed in October 2002 during field activities conducted in support of this BERA.

2.2 Ecological Setting

Sauget Area 2 is located in the floodplain of the Mississippi River in an area known as American Bottoms. Topographically, Sauget Area 2 consists primarily of flat bottomland, although local topographic irregularities do occur. Generally, land surface in the American Bottoms slopes from north to south and from east to west, toward the Mississippi River. Land surface elevation ranges from 400 to 410 feet above Mean Sea Level (MSL) with little topographic relief.

2.2.1 Mississippi River

The section of the Mississippi River adjacent to the Sites is called the Unimpounded Reach (USGS, 1999), which extends from St. Louis downstream to Cairo, Illinois. This section of the river, also called the Open River Reach, is characterized by channelized aquatic habitats with levees to provide flood protection. Area 2 is located approximately two miles downstream of the confluence with the Missouri River. In the vicinity of Area 2, the riverbank has been substantially sculpted by anthropogenic activities. A rock revetment covers approximately 2,250 feet of the riverbank adjacent to Site R, and the remainder of the riverbank has been developed to support barge and shipping traffic. Channel depth in the center of the channel is maintained at a minimum depth of nine feet to allow for barge traffic. The current is swift, with median flows ranging from 85,000 to 95,000 ft^3/second (USGS, 1999).

Observations made during the aquatic sampling in support of this BERA indicate that the depth to bottom of the Mississippi River adjacent to Area 2 varied from approximately 2 feet to approximately 35 feet, with an average water column depth of 16.5 feet. Due to winds and fast moving currents, the water column was well mixed as indicated by the uniform temperature, conductivity, and dissolved oxygen measurements. Temperature values ranged from 7.8 to 9.3° C in bottom water (~one foot above sediment-water interface), mid-depth, and surface water (zero to one foot depth), with an average of 8.6° C (47° F). The pH of surface water was slightly alkaline and ranged from 7.2 to 8.3 in bottom water, 8.0 to 8.5 in mid-depth measurements, and 8.1 to 8.6 in surface water. Dissolved oxygen levels varied from 11.5 to 14.1 mg/L in bottom water, 11.6 to 14.5 mg/L at mid-depth, and 11.6 to 14.4 mg/L in surface water. The bottom sediments exhibited a range of grain sizes, from very fine silty-clay to very coarse gravelly-sand with pebbles (Appendix II-A). The color and organic carbon content also appeared to be highly variable.

Aquatic life within the river depends upon the presence of suitable habitat, which is primarily a function of water and sediment characteristics. Areas of deep, swift water, such as found adjacent to Area 2, would be occupied by channel dwelling fishes and would not support habitat that would be used for spawning or as nurseries. Fremling *et al.* (1989) reported that the Upper Mississippi River Basin supports at least 260 freshwater fish species. Fish in channel habitats are called riverine species and occur as either streamline forms that occupy the water column such as white bass (*Morone chrysops*), or bottom-dwelling forms, such as channel catfish (*Ictalurus punctatus*) (USGS, 1999). Other common riverine species identified by USGS (1999), based on Long-Term Resource Monitoring Program (LTRMP) catch data include sauger (*Stizostedion canadense*), walleye (*Stizostedion vitreum*), and smallmouth buffalo (*Ictiobus bubalus*). Important prey species indigenous to the Unimpounded Reach area include gizzard shad (*Dorosoma cepedianum*) and emerald shiners (*Notropis atherinoides*).

Detailed observations made of the Mississippi River in the vicinity of the Sites can be found in Menzie-Cura (2001) and in the *Aquatic Field Sampling Report* (AMEC, 2003b).

2.2.2 Site P Ecological Conditions

Site P is the northern most of the disposal sites. Site P occupies approximately 20 acres of land and is located between the Illinois Central Gulf Railroad and the Terminal Railroad, north of Monsanto Avenue in the village of Sauget. Figure 2-2 shows the layout of Site P.

Topographically, Site P is approximately 50% flat level ground, with the remaining 50% being part of two topographic gullies, one located in the south-center section of the disposal area and the other laying along the eastern border of the disposal area. In general, the flat areas are dominated by herbaceous vegetation such as Johnson grass (*Sorghum halepense*), white heath aster (*Aster pilosus*), partridge pea (*Cassia fasciculata*), common mullein (*Verbascum thapsus*) and crown vetch (*Coronilla varia*). Occasional early successional hardwood species such as eastern cottonwood (*Populus deltoides*) and smooth sumac (*Rhus glabra*) can be found across these flat areas. More extensive stands of hardwood trees, including eastern cottonwood, white ash (*Fraxinus americana*), American elm (*Ulmus americana*), box elder (*Acer negundo*), and black locust (*Robinia pseudoacacia*), can be found along the sides and across the bottom of the defiles.

The vegetative community throughout Site P is robust, with good vigor and no indication of obvious phytotoxicological effects such as chlorosis, wilting, or mortality. In the vicinity of URS surface soil sampling point W-P-1 (see Figure 2-2), there are fewer plants within the plant community but individual plant growth is vigorous. This appears to be the result of poor soil conditions. Photographs of Site P can be seen in Appendix I-B and in the *Floodplain Field Sampling Plan* (AMEC, 2003a).

2.2.3 SITE R ECOLOGICAL CONDITIONS

Site R is located adjacent to the Mississippi and is covered with a temporary soil cap and is surrounded by a fence. Topographically, this disposal area is flat (see Figure 2-3) and covers approximately 36 acres. The vegetative community is comprised of various herbaceous species and the cap is maintained with mowing. Growth in the community is vigorous and thick. However, because of the mowed condition, it is unlikely that this disposal area supports a significant population of wildlife.

2.2.4 SITE O ECOLOGICAL CONDITIONS

Site O is located on Mobile Avenue in Sauget and occupy approximately 20 acres northeast of the American Bottoms Regional Wastewater Treatment Facility (ABRTF). As noted earlier, Site O contains a subarea, Site O (North), which is the north wing where URS surface soil sample W-O-1 is located. Figure 2-4 shows the layout of Site O.

Topographically, Site O is flat. As the disposal sites are covered with early successional herbaceous species and hardwood species that are relatively young in growth, it appears that these areas have been historically disturbed. Most of Site O (North) is covered by herbaceous species such as Canada goldenrod (*Solidago canadensis*), Johnson grass, crown vetch, and white heath aster. Tartarian honeysuckle (*Lonicera tartarica*), stiff dogwood (*Cornus foemina*), and smooth sumac dominate the tall shrub/scrub community that covers most of Site O. Some larger hardwood species, including eastern cottonwood and slippery elm (*Ulmus rubra*), can be found in both Site O North and Site O. The vegetative community throughout Site O is robust, with good vigor and no indication of obvious phytotoxicological effects such as chlorosis, wilting, or mortality. Photographs of Site O can be can be seen in the *Floodplain Field Sampling Plan* (AMEC, 2003a).

2.2.5 SITE S ECOLOGICAL CONDITIONS

Site S is a small disposal area west-southwest of Site O measuring less than 1 acre in size. As shown in Figure 2-5, Site S is roughly rectangular in shape, with the long sides running approximately north to south. The Site is divided roughly in half with the southern end (containing URS surface soil sample location W-S-2) being entirely paved or covered with gravel. The northern end (containing URS surface soil sample location W-S-1) is vegetated.

The vegetative community in the northern portion of Site S is sharply divided between two distinct community types. One community is an herbaceous community dominated almost exclusively by Johnson grass. The other community is a hardwood forest community containing black locust, eastern cottonwood, American elm, and white mulberry (*Morus alba*). The vegetative communities throughout Site S are robust, with good vigor and no indication of obvious phytotoxicological effects such as chlorosis, wilting, or mortality.

2.2.6 SITE Q ECOLOGICAL CONDITIONS

Site Q occupies approximately 90 acres and is south of Sauget Site R and the old Union Electric Power Plant, west of the Illinois Central Gulf Railroad and the U.S. Corps of Engineers flood control levee, and east of the Mississippi River. Topographically, Site Q (specifically the southern end) is characterized by a greater degree of relief than the other disposal areas (with the possible exception of Site P). The two most significant topographic features are the large pond and small pond, which are two former borrow pits located in the southern end of Site Q that retain water on an intermittent basis.

As previously mentioned, Site Q has been subdivided into Q (North), Q (Central), Q (South), and Q (Ponds). Based on the fact that Q (North) and Q (Central) are covered with pavement, roads, parking areas, debris and buildings, and contained no viable habitat for wildlife, the USEPA agreed that this BERA would focus the assessment on Site Q (South), and Site Q (Ponds).

2.2.6.1 Site Q (South)

Site Q (South) has a significant quantity of river floodplain forest typical to this section of the Mississippi River (see Figure 2-6). The four major floodplain forest communities in the Upper Mississippi River System include those dominated by black willow (*Salix nigra*), those dominated by eastern cottonwood, those dominated by silver maples (*Acer saccharinum*), and those dominated by a mixed oak-hickory forest (USGS, 1999).

Plant species identified by the USEPA in their survey were cocklebur (*Xanthium strumarium*), common mullein, common evening primrose (*Oenothera biennis*), black-eyed susan (*Rudbeckia serotina*) and eastern cottonwood. Identified mammals included eastern cottontail (*Sylvilagus floridanus*) and whitetailed deer (*Odocoileus virginianus*). Identified birds included red-winged blackbirds (*Agelaius phoeniceus*), American robin (*Turdus migratorius*), northern cardinal (*Cardinalis cardinalis*), field sparrow (*Spizella pusilla*), domesticated pigeons (*Columba livia*), American coot (*Fulica americana*), common flicker (*Colaptes auratus*), American kestrel (*Falco sparverius*), and wild turkey (*Meleagris gallopavo*).

Observations made during the October 2002 field event in support of the preparation of this BERA indicated that Site Q (South) supported a more diverse number of vegetative communities than any of the other disposal areas. Generally, Site Q (South) contains two basic types of vegetative communities, herbaceous and hardwood forested. The herbaceous community varies depending upon location, with some locations dominated by a mixed grass and forb community containing Canada goldenrod, white heath aster, Johnson grass, orchard grass (*Dactylis glomerata*), and bushy bluestem (*Andropogon gerardi*). This community can be found in the central portion of the site in the vicinity of URS surface soil sample locations W-Q-11, W-Q-12 and S-Q-13 (see Figure 2-6). Other areas are dominated by common clotbur (*Xanthium chinense*). This community can be found in the vicinity of S-Q-19 and S-Q-20.

The forested community in Site Q (South) shows equal diversity. The forested area surrounding S-Q-16 is dominated by black willow and eastern cottonwood, and contains almost no understory or herbaceous strata. The forest adjacent to W-Q-10 and S-Q-13 is dominated by white mulberry, with a few isolated eastern cottonwoods and no understory or herbaceous strata. The forested community in the vicinity of S-Q-19 and S-Q-20 was dominated by black willow, with an occasional eastern cottonwood. The herbaceous stratum in this area was dominated by common clotbur, with other herbaceous species including jumpseed (*Polygonum virginianum*) and wild cucumber (*Echinocystis lobata*). The forested community in the vicinity of S-Q-17 was dominated by silver maple (*Acer saccharinum*), with an herbaceous stratum in open clearings of tall nettle (*Urtica procera*). S-Q-18 was located in a white mulberry grove.

Photographs of Site Q can be can be seen Appendix I-B and in the *Floodplain Field Sampling Plan* (AMEC, 2003a) and the *Aquatic Field Sampling Report* (AMEC, 2003b).

2.2.6.2 Site Q (Ponds)

The ponds located at the southern end of Q (South) (identified as Q (Ponds)) are ephemeral bodies of water that are subject to cycles of filling and drying, with the length of the cycle being dependent upon rainfall and the flood stage of the Mississippi River. The large pond covers approximately 8.6 acres and the small pond covers approximately 2.6 acres. Both ponds were dry in January 2001, then refilled following flooding of the Mississippi River in March 2002. By August 2002, the small pond was again dry and the large pond dried in December 2002. Both ponds were refilled by stormwater in June 2003.

In November 2002, the large pond was nearly dry, with the water, occupying an estimated 50-foot by 100-foot area. The water column was only 2-6 inches deep and was very turbid. The sediments were sticky clay with silt and contained some organic matter. An herbaceous community that is dominated by hydrophytic plants typical of a wetland surrounded the large pond. Species in this area include umbrella sedge (*Cyperus strigosus*), box sedge (*Carex lurida*), lady's thumb (*Polygonum persicaria*), and water pepper (*Polygonum hydropiper*).

General observations of the large pond, including the types of fish captured during the November 2002 aquatic sampling to support this BERA can be found in *Aquatic Field Sampling Report* (AMEC, 2003b).

2.2.7 SENSITIVE HABITATS

Sensitive habitats include those ecological systems that could support endangered or threatened species (either federally or state listed) or support wetlands. Habitat to support endangered and/or threatened species has not been observed at the Site, which is supported by the lack of endangered or threatened species expected on the Sites (USEPA, 1997b). Menzie-Cura (1999) noted that a pair of bald eagles attempted to nest on the southern end of Arsenal Island, south of the Sites, in 1993. While the pair failed in their first attempt, it is not know whether later attempts were successful. A nest was observed by Menzie-Cura in 1996, but it did not appear to be in use.

A review of the National Wetland Inventory (NWI) map for the Sites, prepared by the U.S. Fish and Wildlife Service (USFWS, 1988), indicated that substantial portions of Sites P and Q were categorized as wetlands at the time the drawings were prepared. These wetlands are listed as palustrine wetlands, dominated by deciduous forests, shrub/scrub plant species, or emergent plant species. Palustrine wetlands are bounded by uplands or any other type of wetlands and may be situated shoreward of lakes, river channels or in floodplains (Cowardin *et al.*, 1979).

Figures showing the location of the NWI identified wetlands for Sites P, R, O, S and the southern end of Site Q are presented as Figures 2-7 through 2-11. It is cautioned that wetlands shown on NWI maps may significantly overestimate the actual amount of land covered by wetlands, due to the basis by which NWI maps are prepared (aerial photointerpretation versus actual field evaluation). Field delineations are required to define the regulatory boundaries of wetlands for permitting purposes. Additionally, most of Site Q (North) and Site Q (Central), as well as northern sections of Site Q (South) have been covered by pavement and buildings, or have been buried under substantial piles of stockpiled material or debris. This physical disturbance is part of ongoing industrial activities managed by one of the property owners.

2.3 Conceptual Site Model

One of the most critical elements of the BERA scoping process is the development of the Conceptual Site Model (CSM). The CSM describes the hypothesized source of COPECs, routes of transport, potential fate mechanisms, potential exposure pathways, and ecological receptors associated with the Sites. The CSM serves as the rationale for the development of sampling plans and protocols, the selection of assessment and measurement endpoints, and the identification of receptors of concern. The CSM can be revised as new site-related information becomes available.

The following sections describe in greater detail the CSM for the aquatic and terrestrial pathways associated with the Sites.

2.3.1 Aquatic Pathways

The Mississippi River is one of the significant pathways associated with the Area 2 Sites and supports a number of important receptors. Figure 2-12 depicts the Aquatic Conceptual Site Model for the Mississippi River. It is important to note that the following discussion assumes a condition where groundwater discharge to the Mississippi River from the disposal areas is uncontrolled. An interim remedy consisting of a barrier wall and extraction wells is being implemented at Site R that will prevent the movement of contaminated groundwater into the Mississippi River.

Two of the five Sites are located in close proximity to the east bank of the Mississippi River (Sites Q and R). The other three Sites (Sites O, P, and S) are located 1500 to 2000 feet east of the riverbank and east of the levee. Solid and liquid industrial and municipal wastes were disposed of on these Sites from the 1950s to the 1980s. At two of the disposal sites, wastes were placed in former borrow pit excavations (Sites Q and R). Wastes were placed in excavations at two other disposal sites (Sites O and S), however, these excavations were made solely for the purpose of waste disposal. Wastes were placed on grade at the fifth disposal site (Site P). It is likely that the excavations at Sites Q and R went to or below the water table to maximize the amount of borrow material. It is unlikely that the excavations for Sites O and S extended to the water table since these disposal sites needed only shallow excavations, 5 to 10 feet deep, to accommodate the materials placed in them.

The aquifer beneath Sauget Area 2 consists of three distinct hydrogeologic units: 1) the Upper Hydrogeologic Unit with fine-grained, silty sands, 2) the Middle Hydrogeologic Unit with clean, medium to coarse sand and 3) the Deep Hydrogeologic Unit with clean, medium to coarse sand and gravel. Leachate migrating from the waste disposal areas could enter these hydrogeologic units and then discharge to the river via groundwater. The ultimate discharge point for these units is the Mississippi River.

COPECs that are discharged to the Mississippi River through groundwater must first pass through the sediments of the river channel prior to entering the water column. In coarse-grained sediments with little organic material, the dissolved groundwater-borne COPECs will pass directly through the sediments and into the water column with only minimal attenuation due to adsorption. In fine-grained or organic rich sediments, a portion of the groundwater-borne constituents may adhere to sediment particles. Whether the constituents remain in the sediment or are dissolved again will depend on their chemical characteristics. Those chemicals with high organic carbon partition coefficient (K_{oc}) values will have a greater affinity for sediment, especially sediment that is high in organic matter. Such constituents would tend to remain sorbed onto sediment particles and migration would occur as a result of sediment movement, not chemical movement.

The primary mechanisms by which chemicals migrate from sediments into the water column are through desorption from sediment particles, resuspension via physical disturbance and resuspension followed by food chain transport. COPECs that are dissolved in groundwater and adsorb onto sediment particles as groundwater wells up through the sediment base may desorb from the sediment particles over time, depending upon the K_{oc} value. For some high K_{oc} constituents, such as PCBs and chlorinated pesticides, desorption from sediment particles, especially those with a high organic content, is very slow, if not minimal. For other constituents, such as polycyclic aromatic hydrocarbons (PAHs) and volatile organic solvents, desorption is much more rapid and can lead to a steady source of the constituent into the water column.

A physical disturbance of the sediment by anthropogenic activities like dredging and prop wash from boats, or natural activities such as flooding can cause resuspension of the sediments followed by desorption of chemicals to the water column. This resuspension may be long- or short-term depending upon the size and solubility of the compound and the size of the sediment particle. Re-suspended particles to which these constituents are sorbed can be either organic matter or inorganic particulates. While in the water column, pelagic flora and fauna may be directly exposed to the re-suspended chemicals as they are transported downstream to other sites. These mobilized constituents in surface water may then be transported through the food chain to higher order trophic levels (i.e., piscivorous and omnivorous wildlife).

Once in the dissolved phase in surface water, the primary migration pathways for chemicals that are transported by groundwater and discharged to surface water would be diffusion throughout the water body. The process of diffusion is an ameliorating process because the compound is reduced in concentration. Diffusion is further enhanced by the flow of water within the river upstream to downstream that moves the diluted chemicals out of the recharge zone.

Another potential migration pathway to the Mississippi River is discharge via storm water runoff. Because Sites O, P, and S are found east of the flood control dike that runs parallel to the river, stormwater runoff would not be a concern. This pathway is not likely to be a major migration pathway at the Sites west of the flood control dike because the areas are covered with vegetation, soil caps, or impermeable pavement. Should it occur, surface water runoff would carry COPECs to the Mississippi River that are either dissolved in the water or adsorbed onto soil or sediment particles. While runoff from Site R is likely limited due to the vegetation over the present cap, there are some areas of Q where runoff may occur. Once in the river, dissolved or suspended COPECs in surface water would be diluted and transported as described above.

The aquatic pathways described in this section also apply to the ponded areas located in the southern portion of Site Q. A conceptual CSM for the ponded areas within Site Q is presented as Figure 2-13. COPECs that are present in the surrounding soils may migrate into the ponds, thereby exposing biota that use, or live, in the ponds. The aquatic community that is present in the ponds is extremely limited and restricted to early successional aquatic plants and some early colonizing benthic invertebrates. Fish are only found in the ponds if a flooding event from the Mississippi River washes them into the ponds. However the ponds may serve as a water source for terrestrial vertebrates and a breeding spot for amphibians or aquatic birds.

2.3.2 TERRESTRIAL PATHWAYS

A CSM for the terrestrial portion of the Sites is presented as Figure 2-14. The migration of COPECs within soils may result in either direct exposure through contact with the soil or indirect exposure through the food chain to faunal communities supported by the available habitat at the Sites. Biota may come in direct contact with chemicals in soil while foraging and/or burrowing. The vectors by which chemicals in the soil may potentially be introduced into biota are direct ingestion (primary source), dermal absorption, or inhalation. The USEPA has determined that inhalation comprises less than 0.1% of total exposure and direct dermal contact comprises approximately 1 to 11% of total exposure (USEPA, 2000a). Indirect exposure occurs when a COPEC is assimilated by a species (e.g., prey/food item) at one trophic level, bioaccumulated by that trophic level, and transferred to the next trophic level through consumption.

Terrestrial receptors may also be exposed to COPECs present in surface water. This pathway is considered to be minor, since most small mammals (particularly rodents) obtain their water through ingestion of plants with high water content, rain puddles collected on the ground or on impervious surfaces, and from dew (Vaughn, 1986). Larger carnivores may drink from ponds, rivers, puddles, or lakes. Terrestrial receptors may also be exposed to COPECs in surface water (and sediment) through either incidental ingestion or through direct contact. Birds may be exposed through the accumulation of soil and sediment by use of these materials as grit.

2.4 IDENTIFICATION OF RECEPTORS OF INTEREST

This BERA utilized an evaluation of representative receptors as indicator species of higher trophic level risks in the aquatic and floodplain environment at the Sites. These receptors of interest (ROIs) were first proposed in the Ecological Risk Assessment Work Plan (Section 12.0 of the SSP). The following ROIs were selected for use in preparing this BERA:

For the Mississippi River aquatic community:

- The benthic macroinvertebrate community;
- Fish (Gizzard shad, channel catfish, and drum);
- Mink (*Mustela vison*); and
- Osprey (*Pandion haliaetus*).

Benthic invertebrates were selected as ROIs because they have the greatest exposure to bottom sediments that potentially contain COPECs, and they are an important link in the aquatic food chain as a food source for bottom feeding fish species in the river. Fish were selected as ROIs because they are the dominant organisms in the water column and they may be exposed to COPECs in sediments and surface water. Fish represent a food source to higher order predators (both aquatic and semi-aquatic) and are important for both recreational and commercial anglers. The species evaluated include water-column and bottom-dwelling species.

The mink and osprey were selected as upper trophic level ROIs. Both species are either found in the area, or have the potential for being found in the area. They both feed on fish, so they can be tied via the food web to the sediments and surface water of the Mississippi River. Additionally, both species are sensitive to constituents that biomagnify up the food web. From a habitat standpoint, the riverbank adjacent to the Sauget Area 2 Sites is not good habitat for any fish-eating mammal. Much of the 14,000 linear foot riverbank is covered with stone riprap, removing cover requirements that the animal has. The remainder of the bank contains pier, pilings, buildings and other human disturbances, which would further preclude mink from inhabiting the area.

For the floodplain terrestrial community:

- Prairie vole (*Microtus ochrogaster*);
- Short-tailed shrew (*Blarina brevicauda*); and
- Red fox (*Vulpes vulpes*).

The prairie vole was selected as a ROI because it is likely to be the dominant herbivore within the habitat provided by the Sites and, with its small home range, could likely spend its entire life span within a Site. Shrews were chosen as a ROI because a large portion of their diet consists of earthworms that live within the soils of the Sites. However, the habitat afforded by the majority of the disposal sites is not particularly supportive of short-tailed shrews (Ballenger, 2000). The red fox is an upper trophic level carnivore that potentially feeds on either shrews or other small rodents within the Sites.

Relative to the Site Q (Ponds), realistic ROIs cannot be selected as the two ponds are ephemeral in nature and do not support a long-term ecological community. Instead, the ponds cycle back and forth between supporting an early emergent aquatic community and supporting a terrestrial community. The ponds only support a fish community following overbank flooding of the Mississippi River. The community would become greatly distorted as the ponds shrink in size due to drying. The loss of habitat and the inability for new individuals to be recruited into the ponds would result in a population dominated by large, scavenging species that are tolerant of diminishing water quality conditions. When the ponds dry up, the fish communities are lost.

2.5 SELECTION OF ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment endpoints are statements of the characteristics or attributes of the environment that are to be protected. These assessment endpoints were identified and are discussed further in the USEPA-approved Ecological Risk Assessment Work Plan (Section 12.0 of the SSP). The following assessment endpoints were defined for the BERA:

- Assessment Endpoint #1: Evaluate the potential for adverse changes in the survival, reproduction, and growth of fish populations utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in sediments, surface waters, and/or prey;

- Assessment Endpoint #2: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of piscivorous wildlife utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in prey;

- Assessment Endpoint #3: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of vermivorous wildlife utilizing the six Sites resulting from exposures to COPECs in prey and in soil;

- Assessment Endpoint #4: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of herbivorous wildlife utilizing the six Sites resulting from exposures to COPECs in soils and/or vegetation;

- Assessment Endpoint #5: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of carnivorous wildlife utilizing the six Sites resulting from exposures to COPECs in prey; and
- Assessment Endpoint #6: Evaluate the potential for adverse changes in the survival, reproduction, and growth of potential fish populations within the two ponds resulting from exposures to COPECs in surface waters and sediments.

The BERA has evaluated ecological risks relative to these assessment endpoints in the Mississippi River and the five disposal sites. Since assessment endpoints cannot be measured directly, measurement endpoints have been identified. Measurement endpoints typically have specific strengths and weaknesses related to the factors discussed above. Therefore, it is common practice to use more than one measurement endpoint to evaluate each assessment endpoint. These measurement endpoints are discussed further in the USEPA-approved Ecological Risk Assessment Work Plan (Section 12.0 of the SSP). The specific measurement endpoints that were used in this BERA are as follows:

Assessment Endpoint #1: Evaluate the potential for adverse changes in the survival, reproduction, and growth of fish populations utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in sediments, surface waters, and/or prey.

Measurement Endpoints for Assessment Endpoint #1

a. The first measurement endpoint was the comparison of COPEC concentration data obtained through the chemical analysis of sediments appropriate sediment quality benchmarks for the protection of benthic macroinvertebrates. The focus was the ability of the benthic community to perform its role as a prey base for fish.

b. The second measurement endpoint was the evaluation of sediment bioassay data.

c. The third measurement endpoint was the comparison of identified concentrations of COPECs from all sampling locations to relevant ecologically based threshold benchmarks, including the State and Federal Ambient Water Quality Criteria for the protection of aquatic life.

d. The fourth measurement endpoint was the evaluation of surface water bioassay data. The ability of surface water to potentially support viable populations of fish was assessed by evaluating survival rate test data from bioassays conducted on surface water samples collected upstream, downstream and in the vicinity of the Sites.

e. The fifth measurement endpoint was the evaluation of whole body COPEC concentrations identified in fish collected upstream, downstream and in the vicinity of the Sites. The fish tissue residue concentrations used for this line of evidence were collected and analyzed by Menzie-Cura as part of the Krummrich work (Menzie-Cura, 2001). The body burden levels were compared to tissue residue data from the literature (e.g., Jarvinen and Ankley, 1999) that indicates potential ecological concerns.

<u>Assessment Endpoint #2:</u> Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of piscivorous species utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in prey.

Measurements Endpoints for Assessment Endpoint #2

a. Potential risks to mink were estimated by comparing an estimated average daily dose for each potential COPEC to a toxicity reference value (TRV) for each potential COPEC. Exposure concentrations to mink were estimated using a food chain model with fish COPEC body burdens from the Mississippi River and from the large pond.

b. Potential risks to ospreys were estimated by comparing an estimated average daily dose for each potential COPEC to a TRV for each potential COPEC. Exposure concentrations to ospreys were estimated using a food chain model with fish COPEC body burdens from the Mississippi River. Fish from the large pond were not used in the exposure calculation due to the size and shallow depth of the pond that would preclude its use by osprey as a feeding area.

Assessment Endpoint #3: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of vermivorous wildlife resulting from exposures to COPECs in prey.

Measurements Endpoints for Assessment Endpoint #3

a. Potential risks to short-tailed shrews were estimated by comparing an estimated daily dose for each potential COPEC to TRVs identified in the literature. Exposure concentrations to short-tailed shrews were estimated using a food chain model and earthworm COPEC body burdens developed through earthworm bioaccumulation tests and terrestrial invertebrate COPEC body burdens collected through field studies in support of this BERA. It is believed, based on habitat observations made during the fieldwork conducted in support of this BERA, that much of the habitat in the disposal sites is not conducive for the presence of populations of short-tailed shrew (Ballenger, 2000). However, the value of this endpoint is in the estimation of hypothetical risks to higher trophic level organisms as measured in measurement endpoint #5. As such, the shrew is considered more as a transfer mechanism to higher trophic level organisms, as opposed to an endpoint receptor.

Assessment Endpoint #4: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of herbivorous wildlife resulting from exposures to COPECs in soils and/or vegetation.

Measurements Endpoints for Assessment Endpoint #4

a. The ability of the plant community to provide habitat for herbivorous wildlife was measured by the comparison of concentrations of COPECs in surface soils at the Sites to surface soil quality benchmarks for the protection of plants.

b. Potential risks to prairie voles were estimated by comparing an estimated daily dose of each potential COPEC to TRVs identified in the literature. Exposure concentrations to prairie voles were estimated using a food chain model and plant COPEC tissue concentrations developed from plant samples taken from the various sites.

Assessment Endpoint #5: **Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of carnivorous wildlife utilizing resulting from exposures to COPECs in prey.**

Measurement Endpoints for Assessment Endpoint #5

a. Potential risks to the red fox were estimated by comparing estimated daily dose of COPECs to TRVs identified in the literature. Exposure concentrations to red fox were estimated using a food chain model and estimated COPEC body burdens in prairie voles and shrews.

Assessment Endpoint #6: **Evaluate the potential for adverse changes in the survival, reproduction, and growth of potential fish populations within the two ponds located in the southern end of Site Q resulting from exposures to COPECs in surface waters and sediments.**

Measurement Endpoints for Assessment Endpoint #6

a. The same measurement endpoints used for the Mississippi River were also used for evaluating the viability of a potential aquatic community within the ponds (endpoints 1a through 1e. Because only the large pond contained water and a fish population at the time of the BERA ecological field support activities, only that pond was evaluated in this BERA. Additional surface water and sediment samples were collected from the large and small ponds in June 2003. This data was collected too late to be considered in this BERA and will be presented at a future date as an addendum to this BERA.

Section 3

3.0 AQUATIC ECOLOGICAL EXPOSURE ASSESSMENT

This section describes the data used in the development of the aquatic assessment and the selection of the aquatic COPECs evaluated in this BERA.

3.1 DATA USED IN THE AQUATIC ECOLOGICAL ASSESSMENT

All chemical data used in this BERA were collected in November 2002 specifically for this project, except for the fish tissue samples, which were collected by Menzie-Cura & Associates from October through November 2000 as part of the baseline ecological risk assessment for the aquatic habitat adjacent to disposal Site R and near the W. G. Krummrich Plant in Sauget, IL (Sauget Area No. 1; Menzie-Cura Associates, 2001).

The chemical data collected in November 2002 for the BERA were collected following the *Aquatic Field Sampling Plan* (URS, 2002) and subsequent modifications as described in the *Aquatic Field Sampling Report* (AMEC, 2003b). AMEC (2003b) also provides detailed information on aquatic sampling locations, sampling methods, and quality assurance/quality control (QA/QC) sample collections.

3.1.1 SAMPLING LOCATIONS

Sediment and water samples were collected from six sampling areas (labeled R1, R2, R3, R4, R5, and R6), located along some 14,000 ft of the Mississippi River (Figure 3-1), with R1 being the most upstream area and R6 the most downstream area. Each sampling area consisted of seven sampling locations, three located on a transect approximately 50 feet from the riverbank (the "A" transect), three more on a transect approximately 150 feet from the riverbank (the "B" transect"), and one located about 300 ft from the riverbank (the "C" transect), for a total of 42 locations. The extensions "U", "M" and "D" on the sampling locations refer to upstream, middle and downstream portions of the transects, respectively. The water column depth at these sampling locations varied from about 2 feet to 32 feet. The sampling locations and their GPS coordinates are summarized in Table 3-1.

The R1 plot area was located upstream of Site P (of Sauget Area 2 Sites), between the MacArthur Railway Bridge and Interstate Highway Bridge (Figure 3-1). Data collected from the R1 plot area are used as "reference" values for the Mississippi River. Data from all other sampling locations (areas 2 - 6) are treated as "Site" related.

As described earlier, fish tissue sampling was not conducted in this study. Instead, fish tissue data collected by Menzie-Cura & Associates (Menzie-Cura, 2001) were used. Fish samples were collected from the portion of Mississippi River adjacent to Site R extending approximately 300 feet into the river channel. The Site area, referred to as the PDA (Plume Discharge Area), is located immediately downstream of the old power plant and situated between sampling plots R2 and R3 in this study. In addition to the PDA, Menzie-Cura collected fish samples from two other areas: an upstream reference area, referred to as UDA (Upstream of Discharge Area) situated adjacent to Site P, in the area between sampling plots R1 and R2 in this report and a downstream reference area, referred to as DDA (Downstream of Discharge Area), is situated downstream of plot area R6 (Figure 3-1).

3.1.2 Sediment and Surface Water Sampling

Field characterization of sediment properties included visual observation of grain size, presence/absence of organic matter, and presence/absence of any benthic organisms. Water quality parameters (pH, temperature, conductivity, dissolved oxygen, and turbidity) were collected at each sampling location using a Horiba (Model U-22) water quality instrument. Water quality parameters were measured near the bottom (approximately one foot above the sediment layer), at mid-depth, and at the surface. These field observations are summarized in Tables 3-2 and 3-3, respectively.

Sediment samples were collected for the analysis of grain size, Total Organic Carbon (TOC), volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), pesticides, herbicides, PCBs, dioxins (at selected locations) and metals. Grain size results are presented in Appendix II-A. Bulk sediment samples were also collected for chronic bioassay (toxicity) testing. Laboratory results for sediments are provided in Appendix II-B.

Surface water samples were collected for the analysis of hardness, VOCs, SVOCs, pesticides, herbicides, PCBs, dioxins (at selected locations), and dissolved and total metals. Surface water samples were also collected for acute and chronic bioassay (toxicity) testing. Laboratory results for surface water are provided in Appendix II-C.

The revised Field Sampling Plan (Addendum to Volume 3 of the RI/FS support sampling plan, AMEC 2003b) called for sampling of sediments for benthic invertebrate community structure analysis only if field observations indicated that the substrate was substantially different from those sediments observed in the Sauget Area 1 work. Based on this, sediment samples were collected for benthic community structure analysis only at two locations, one from the Site (plot area 6) and one from the reference area for comparison purposes. This activity was not listed as a line of evidence in the Ecological Risk Assessment Work Plan (Section 12.0 of the SSP), and is being used on a qualitative basis within this BERA.

3.1.3 TOXICITY TESTING

Bioassays are direct measures of the relative toxicity of constituents in a particular matrix. The ability of the benthic community to potentially perform its role as a prey base for fish was assessed by evaluating the survival rates of bioassay test organisms following exposure to sediments from the Site and reference areas. Sediments were subjected to the following bioassays:

- *Hyalella azteca*, 10-day chronic survival and growth test
- *Chironomus tentans*, 10-day chronic survival and growth test

Sediment bioassays were conducted in accordance with U.S. Environmental Protection Agency protocols outlined in "Methods for Measuring the Toxicity and Bioaccumulation of Sediment-associated Contaminants with Freshwater Invertebrates, Second Edition" (USEPA, 2000b) and with American Society for Testing and Materials (ASTM) protocols outlined in "Standard Guide for Conducting Sediment Toxicity Tests with Freshwater Invertebrates" (ASTM, 1994). The protocols are presented in Appendix II-D. The results of the sediment bioassays are presented in Appendix II-E.

The ability of surface water to potentially support viable populations of fish was assessed by evaluating the survival rate of test organisms. The following surface water bioassays were conducted in support of this BERA:

- *Pimephales promelas*, acute (4-day) and chronic (7-day) growth and survival test
- *Ceriodaphnia dubia*, acute (2-day) and chronic (7-day) survival and reproduction (neonate production) test

P. promelas and *C. dubia* bioassays were conducted in accordance with USEPA protocols outlined in "Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms" (USEPA, 1989b). The protocols are presented in Appendix II-C. The results of the surface water bioassay testing are presented in Appendix II-F.

3.1.4 Bioaccumulation Testing

Bioaccumulation tests were conducted on Site sediments as described in the Field Sampling Plan. These results were used as one line of evidence together with toxicity test results and chemical analysis data. Bioaccumulation tests were run for 28 days using *Corbicula fulminea* (Asiatic clam). At the end of the experimental period, surviving clams were frozen and shipped to the analytical laboratory for the analysis of target analytes. Detailed experimental procedure and methods are given in Appendix II-C. Results of chemical analysis of clams from bioaccumulation testing are given in Appendix II-G.

3.1.5 Fish Sampling

As described earlier, fish tissue collection was not conducted in the Mississippi River for this study. Fish collected by Menzie-Cura for the Krummrich ecological risk assessment was used for this report. Details on fish sampling procedures can be found in Menzie-Cura Associates (2001) report. A brief summary is provided here.

Gizzard shad, channel catfish, and drum were collected by Menzie-Cura for the Krummrich ecological risk assessment. These warm water fish species were selected because they a) reflect local sediment and water quality conditions since the species include both bottom-dwelling and water-column feeders; b) are abundant local residents with a limited foraging area; c) can be exposed to sediment as well as surface water; d) represent fish and higher order predators that feed on smaller fish and invertebrates; and e) can serve as a prey base for avian and mammalian species. The fish tissue data obtained from Menzie-Cura is presented in Appendix II-H.

3.1.6 DATA VALIDATION, DATA QUALIFIERS AND USE OF DATA

Data validation on all chemical analysis was conducted by URS Corporation (URS, 2003b). The data validators applied the following qualifiers where warranted:

J The associated numerical value is an estimated quantity due to quality control exceedance(s). The value is used for project decisions as an estimated result.

U The compound was analyzed for, but was not detected. The associated numerical value is the sample detection/quantitation limit. One-half the reporting limit was used for the estimation of average concentrations.

UJ The compound was analyzed for, but was not detected. The associated numerical value is the sample detection/quantitation limit, and is an estimated quantity. One-half the reporting limit was used for the estimation of average concentrations.

R This data was rejected due to severe or cumulative exceedance of quality control criteria. The value was not used for project decisions and was excluded in all statistical analysis.

The data validators recommend that, due to low precision, the results of acetone and bromomethane be used with caution when using the data for decision-making. There were no other major data quality issues.

3.1.7 CALCULATION OF PCB AND DIOXIN/FURAN CONCENTRATIONS

Surface water, sediment, and fish samples were analyzed for PCB homologs, polychlorinated dibenzo-p-dioxin (PCDDs) and polychlorinated dibenzofuran (PCDFs) congeners. PCBs, PCDDs and PCDFs are complex mixtures of individual congeners that exhibit different solubilities, volatilities, degradation, metabolism, and toxicities.

PCBs were only detected in sediments, and were not detected in surface water, bioaccumulation test organisms (clams) or in fish tissue. At least one PCB homolog was detected in five sediment samples from the following locations: four from Plot R3 (locations R3AD, R3AM, R3AU, and R3BM), and one from Plot R4 (R4BM). In these samples, the concentrations of individual homologs (or one-half the detection limit for a non-detect) were summed up on a sample-by-sample basis. If a homolog was never detected in any of the sediment samples, then it was not included in the total.

Dioxins and furans were collectively evaluated as dioxin Toxic Equivalent (TEQ) concentrations in various media. Of the various dioxins/furans that bind to an intracellular protein called the aryl hydrocarbon receptor (AhR), 2,3,7,8-TCDD is the most toxic form. The toxicity of other congeners of dioxins and furans were compared to the toxicity of 2,3,7,8-TCDD by the application of Toxicity Equivalency Factor (TEF) for fish developed by the World Health Organization (WHO). TEQ concentrations in samples were calculated using the following equation (Van den Berg *et al*., 1998):

$$TEQ = \sum_{n1}[PCDD_i \times TEF_i] + \sum_{n2}[PCDF_i \times TEF_i]$$

where,

TEF = the toxic equivalency factor for dioxin (PCDD) or furan (PCDF) congener "i"
n = number of dioxin-like congeners detected in a given medium.

TEQs for dioxin/furans were calculated for each medium by multiplying the detected concentration (or one-half the detection limit for a non-detect, as appropriate) of each individual congener by its TEF and adding the products to obtain the TEQ. If a congener was never detected in a particular medium or area, it was not included in the total. The TEQ concentration was then compared to appropriate screening numbers and threshold values. Calculated TCDD-TEQs are presented in Table 3-4.

3.2 DATA ANALYSIS AND AQUATIC COPEC SELECTION

The results from the chemical analyses for sediments and surface water were used to develop the list of COPECs for the aquatic environment. The sample locations and number of proposed samples were developed during discussions with the USEPA and were biased to reflect potential worst-case conditions in the aquatic environment.

COPECs in sediments and surface water were identified through a multi-step process. Step 1 included the screening of chemical data obtained through sediment and surface water sampling. As per the USEPA-approved work plan, the initial screening of sediment and surface water data included:

- Comparison to Background – The BERA eliminated from further analysis a constituent that occurred below the maximum concentration measured the reference area for a given medium.
- Frequency of Detection – The BERA eliminated a constituent from the evaluation if the constituent was detected in less that 5% of samples from a particular medium.

If the constituent was not eliminated through this initial screening process, it was carried through the full risk assessment process. After this initial screening process, 34 analytes were retained as COPECs in surface water. Fifty-five analytes were retained as COPECs in sediment. These initial sediment and surface water COPECs are listed in Tables 3-5 and 3-6 for sediment and surface water, respectively.

The following sections describe the procedures used for analyzing data to meet the previously described measurement endpoints.

3.2.1 ANALYSIS OF SEDIMENT AND SURFACE WATER CHEMICAL DATA

Sediment and surface water data from the reference area (R1 plot area) data were analyzed separately from the Site area (plot areas R2 through R6) data. For all analytes carried through the initial screening process, maximum, arithmetic average, and frequency of detection were calculated for the reference area and the Site area. A summary of these calculations is provided in Tables 3-7 and 3-8 for sediments and surface, respectively.

Concentrations of COPECs in surface water and sediment were compared to relevant ecologically based threshold benchmarks (described in Section 4.1) using comparison protocols outlined in Section 3.2.

3.2.2 Analysis of Sediment Toxicity Data

Sediments were subject to toxicity tests using two invertebrate species, *H. azteca* and *C. tentans*. The endpoints of the tests were survival and growth. Sediment bioassay samples were grouped in the laboratory by sampling area. In addition, each group was run with a laboratory control sample. Survival and growth data were recorded for the samples and the controls by the laboratory. Survival data were reported as percent survival, whereas growth is recorded as average mass per specimen. Summary results of the sediment toxicity tests using *H. azteca* and *C. tentans* are shown in Table 3-9, and the complete result of statistical tests are provided in Appendix II-E.

Prior to statistical analysis, all percent survival data were transformed using an arcsine-square root transformation, consistent with the approach taken in the Krummrich ERA that allows the application of parametric statistical tests. To determine whether significant mortality occurred for field samples when compared to their respective laboratory control sediment samples, mean survival for each field sample was compared to the mean survival of their respective laboratory control sample using Dunnett's one-way ANOVA. An alpha of 0.05 was used to assess the statistical significance. The null hypothesis, H_o, was that the mean survival of each of the field samples was equal to the mean survival of the laboratory control sample.

In addition to survival, comparison of mean growth levels was conducted using the Dunnett's one-way ANOVA at an alpha of 0.05. The null hypothesis, H_o, was that the mean growth of each field sample was equal to the mean growth of the laboratory control sample. Statistical analysis of growth was conducted using the raw, untransformed data.

3.2.3 Analysis of Sediment Bioaccumulation Test Data

Bioaccumulation tests are designed to evaluate the bioaccumulation potential of constituents of concern in the tissue of sediment-dwelling organisms. While not considered as an actual measurement endpoint, this evaluation can be used to assess the potential for COPECs in sediment to accumulate in tissue.

Chemical analysis results of clam tissue from bioaccumulation tests are provided in Appendix II-F. For all target analytes, maximum, arithmetic average and frequency of detection were calculated for Site samples (plot areas R2 through R6), reference samples (plot area R1), and controls. A total of 32 target analytes were detected in Site samples and are summarized in Table 3-10.

Chemical data from clam tissue was compared to the values presented in the web-based database called *Environmental Residue Effects Database (ERED)* developed by the US Army Corps of Engineers (2002) as a means of determining potential impacts to bivalves that might exist in sediments found in the river adjacent to the Area 2 Sites. The results of that comparison were then compared to fish tissue body burdens determined by Menzie-Cura in a qualitative manner. That comparison provided insight into the accumulation of COPECs from the sediments up through the aquatic food chain.

3.2.4 Analysis of Surface Water Toxicity (Bioassay) Data

As described earlier, surface water toxicity tests were conducted using *P. promelas* and *C. dubia.* Endpoints assessed were survival and growth (mean weight per fish). For *C. dubia* the endpoints assessed were survival and neonate production. Samples were grouped by sampling area and analyzed separately. Each grouping of samples was evaluated against its respective laboratory control sample. Results of the chronic bioassay for the *P. promelas* and *C. dubia* are summarized in Table 3-11.

<u>Fathead Minnows</u>: To assess whether significant toxicity was present during the bioassay of fathead minnows, statistical testing was conducted using a tiered approach. This approach was designed to identify significant differences in survival first between individual samples and their

respective laboratory control samples, second between individual samples and other intra-area samples, and third between individual samples and other inter-area samples. The goal of the statistical analysis was to identify spatial patterns in toxicity that could be compared to corresponding exposures (surface water COPEC concentrations), which ultimately yields an exposure/response relationship. Statistical tests and results are summarized in Appendix II-D.

The first tier of the assessment was to determine whether significant mortality occurred for individual field samples when compared to their clean, laboratory control water samples. To assess this, the mean survival for field samples was compared to the mean survival of their respective laboratory control sample using Dunnett's one-way Analysis of Variance (ANOVA). An alpha of 0.05 was used to assess the statistical significance. Prior to running the ANOVA, all percent survival data was transformed using an arcsine square root transformation. The null hypothesis, H_o, was that the mean survival of each field sample was equal to the mean survival of the laboratory control sample.

In addition to survival, growth data from field samples were compared to growth data from their respective laboratory control results. This analysis was also conducted using the Dunnett's one-way ANOVA with an alpha of 0.05. The results of this analysis were used to assess whether there were significant differences in mean growth between laboratory control samples and field samples. In this case, the null hypothesis, H_o, was that the mean growth of each field samples was equal to the mean growth of the laboratory control sample.

If the result of the Dunnett's one-way ANOVA for either survival or growth indicated that field sample results were significantly different (lower) than their respective laboratory control (i.e., the field samples were experiencing higher mortality or lower growth) then a Tukey's pair wise ANOVA with a family error rate or alpha of 0.05 was run to determine if there were significant differences in mean mortality and/or growth between individual sample location pairs.

<u>Ceriodaphnia</u>: To assess whether significant toxicity was present during the bioassay of *C. dubia*, test results were subject to a tiered approach similar to that described above for fathead minnows. The results of the statistical analyses for *C. dubia* are provided in Appendix II-D. The first tier of the assessment was to determine whether significant mortality to *C. dubia* occurred for field samples when compared to the clean, laboratory control water samples. To assess

this, the mean survival for field samples was compared to the mean survival of the laboratory control sample using Dunnett's one-way ANOVA. For the *C. dubia* survival analysis, control sample results were pooled because examination of those data indicated no significant differences between control survival results based on Tukey's pair wise ANOVA at an alpha of 0.05. An alpha of 0.05 was used to assess the statistical significance. The null hypothesis, H_o, was that the mean survival of each of the field samples was equal to the mean survival of the laboratory control sample.

In addition to survival, *C. dubia* neonate production data from field samples were also compared to their respective laboratory control results. This analysis was conducted using the Dunnett's one-way ANOVA with an alpha of 0.05. The results of this analysis were used to assess whether there were significant difference in neonate production between laboratory control samples and field samples. In this case, the null hypothesis, H_o, was that the neonate production of each field sample is equal to the mean neonate production of the laboratory control sample.

If the result of the Dunnett's one-way ANOVA for either survival or neonate production indicated that field sample results were significantly lower than the laboratory controls, then a Tukey's pair wise ANOVA with a family error rate or alpha of 0.05 was run to determine if there were significant differences in mean mortality and/or neonate production between individual sample location pairs.

3.2.5 Analysis of Fish Tissue Data

Exposure levels in fish were evaluated by comparing the concentrations of COPECs in fish tissue collected from the Site to the levels in fish from reference areas. Care needs to be exercised with such a comparison because discriminating between Site-related and non-Site-related chemical sources is difficult in wide ranging species such as fish.

Fish tissue data from the Krummrich report are summarized in Appendix II-H. As described earlier, the data used in this risk assessment include whole body composites of drum, gizzard shad, and channel catfish from the plume discharge area (Site), upstream reference area (UDA) and downstream reference area (DDA).

For all target analytes, maximum concentration, arithmetic average concentration and frequency of detection were calculated for Site samples (PDA) and reference areas (UDA and DDA). All fish species and composites from the Site area were aggregated together for data analysis. Similarly, UDA fish data were analyzed together and DDA fish data were analyzed together. The results are summarized in Table 3-12.

None of the PCB homologs were detected in any of the fish tissue. Concentrations of all detected dioxin congeners were converted to toxic equivalent (2,3,7,8-TCDD TEQ) concentrations as described in Section 3.1.8 (Table 3-13). A total of 19 target analytes (all dioxins/furans treated as one analyte, 2,3,7,8-TCDD TEQ) were detected in fish tissue and are summarized in Table 3-14. Concentrations of two compounds (2-methylphenol and 4,4'-DDE) were higher in downstream reference samples than in Site fish samples. For all other analytes, they were either not detected in reference samples or were below the maximum concentrations found in Site fish tissue. Although dioxin/furans are evaluated using a TEQ approach, concentrations of several individual dioxin congeners were higher in upstream (UDA) and/or downstream (DDA) reference area fish than in the Site (PDA) area fish. However, the 2,3,7,8-TCDD TEQ concentrations were slightly higher in Site area samples than in reference area samples.

Section 4

4.0 FLOODPLAIN ECOLOGICAL EXPOSURE ASSESSMENT

This section presents the concentrations of COPECs that receptors within the floodplain community may be exposed to. These exposure concentrations include measured values in surface soil of the disposal sites, pond surface water and sediment concentrations, measured values identified in fish tissue collected from the large pond, and modeled calculations of exposure to higher trophic level organisms. Data used in the Floodplain Assessment portion of this BERA was collected by URS Corporation in the summer of 2003 and by AMEC Earth & Environmental in October 2003. The *Field Sampling Report* (URS, 2003a) and *Floodplain Area Field Sampling Report* (AMEC, 2003a) provide details regarding the sampling procedures and activities. Sampling activities for the pond are summarized in *Aquatic Field Sampling Report* (AMEC, 2003b).

The floodplain community evaluated in this BERA also includes the two ponds located at the southern end of Site Q and the aquatic community supported by them. These ponds are intermittent in nature and cycle through wet and dry periods. The two significant contributors of water for these ponds are overbank flooding of the Mississippi River and rainfall. Both ponds were dry in December 2002. Limited sampling of surface water, sediment and fish was possible from the large pond in November 2002. Consequently, the data from the ponds evaluated in this BERA reflect transient conditions that may not be representative of current conditions. Supplemental sampling of the ponds occurred in June 2003 after the ponds had refilled. That data will be evaluated in an addendum to this BERA.

4.1 SURFACE SOIL SAMPLING RESULTS

Concentrations of chemical constituents present in the surface soil of the various disposal sites were identified through the collection and laboratory analysis of surface soil samples. Surface soil samples from the disposal sites were collected by URS Corporation during the remedial investigation fieldwork in July and August 2002. Samples from off-site locations used as background information were collected at the same time. Additional soil samples were collected by URS Corporation in October 2002. Samples were collected from the top six inches of the soil matrix. Based on an agreement with the USEPA, soil samples collected in the northern and central sections of Site Q were not evaluated in the BERA because of a lack of habitat in that portion of the Site. Only the southern end of Site Q has sufficient habitat to support a diverse ecological community.

Four surface soil samples were collected from Site P (Figure 2-2). Four surface soil samples were collected from Site R (Figure 2-3). Three soil samples were collected in Site O (Figure 2-4). Soil sample W-O-1 is located in a subarea of Site O identified as Site O (North). One surface soil sample was collected in Site S (see Figure 2-5) in an area that afforded ecological habitat. Four surface soil samples were originally collected in Site Q. However, as a result of comments made by the USEPA during the work plan development, eight additional surface soil samples were collected from the southern end of Site Q in October 2002 as part of the fieldwork in support of the floodplain assessment. Figure 2-6 shows the location of all twelve surface soil samples in the southern section of Site Q. In addition to the site-related samples, off-site samples were collected to provide data regarding the characteristic concentrations of COPECs in non-site related soils. Three off-site locations were used for comparison purposes in this BERA (OS-2, OS-3, and OS-4). Sample locations are shown on Figure 2-1.

Surface soil samples were generally biased towards areas of known contamination. The number of soil samples was first agreed to by the USEPA. Then prior to the collection of soil samples during the RI field sampling activity, URS Corporation conducted a reconnaissance survey of the disposal sites using field screening techniques including analysis using a mobile laboratory and soil gas surveys. The location of the surface soil samples was made based on those results.

Elevated concentrations of a wide variety of organic and inorganic constituents in comparison to the off-site locations were found in the surface soil of the disposal sites. In general, Site R had the lowest overall concentrations of constituents in the surface soil. Noting the temporary cap that was placed over this disposal area, the lower concentrations were not unexpected. W-O-1 generally had the highest overall concentrations of volatile organic compounds (VOCs), polychlorinated biphenyls (PCBs), and dioxin in surface soil compared to off-site sampling areas. Site S generally had the highest overall concentrations of semi-volatile organic compounds (SVOCs) and pesticides. The southern end of Site Q generally had the highest overall concentrations of inorganic constituents and herbicides.

Surface soil data are presented and evaluated on a per sample site-by-site basis for the majority of the ROIs that were evaluated (with the exception of the red fox). Additionally, an average concentration for each analyte in the surface soil data in each disposal area was calculated. For the red fox, a 95% UCL and an average concentration was calculated for each analyte on a site-wide basis. Appendix III-A (Table III-A1) presents the surface soil data used in the ecological risk assessment. A more extensive presentation of the data can be found in the *Sauget Area 2 Data Report* (URS, 2003b).

4.2 Pond Surface Water Sampling Results

Two surface water samples were collected from the large pond in November 2002. At that time, less than 10% of the bottom of the pond basin contained water and the small pond had entirely dried up. Low concentrations of organic and inorganic constituents were identified in the surface water samples. Several dioxin/dibenzofuran congeners and metals were detected in these samples. Appendix III-A (Table III-A2) presents the pond surface water data used in the ecological risk assessment.

As a result of high rainfall events, both ponds had refilled with water in May 2003. In June 2003, three surface water and sediment samples were collected from each of the two ponds and analyzed for the standard suite of chemicals, and bulk samples were taken for performance of bioassay tests. The results from those analyses will be submitted as an addendum to this BERA.

4.3 Pond Sediment Sampling Results

One sediment sample was collected from the large pond in November 2002. Elevated concentrations of PCBs, dioxins, and metals were identified in the sample. Appendix III-A (Table III-A3) tabulates the pond sediment data used in the ecological risk assessment.

4.4 Plant Tissue Concentrations

To provide site-specific data regarding the potential exposure of COPECs that have been assimilated by terrestrial plants to herbivorous wildlife feeding on those plants, plant tissue was collected from the five disposal sites. A total of 24 plant tissue samples were collected from onsite areas and three samples were collected from off-site (background) areas. The sampling procedures were presented in the *Floodplain Area Field Sampling Report* (AMEC, 2003a). Plant tissue concentration results are summarized in Table III-A4.

Relatively few SVOCs were found in the site related plant samples in comparison to the off-site (background) samples. This was expected as plants generally do not carry many of the SVOCs (particularly the polycyclic aromatic hydrocarbons) or they tend to metabolize them (see Appendix III-E). Concentrations of pesticides and herbicides, as well as dioxin/furans were identified in plant tissue. The greatest detections were of inorganic constituents. Inorganic are integral components of plant tissue (Pandas and Pandas, 1992, and Allen *et al.*, 1995), so that result was expected.

4.5 Earthworm Tissue Concentrations

To provide site-specific data regarding the potential exposure of COPECs that have been accumulated by earthworms that are fed upon by vermivorous wildlife such as short-tailed shrews, 28-day earthworm bioaccumulation tests were conducted using soil collected from the disposal sites. Following completion of the bioaccumulation tests, the test earthworms were analyzed for COPEC tissue residue concentration residues. Earthworm bioaccumulation tests were conducted at the 24-onsite sampling locations and the 3 off-site sampling locations. Earthworm tissue concentration results are summarized in Table III-A5.

In comparison to off-site sampling locations, a variety of organic and inorganic constituents were identified in earthworm tissue from the earthworm bioassay tests. PCBs and dioxins/furans were identified, with the highest concentrations being from Site O and Site S. The highest concentrations of herbicides and pesticides were identified from Site P, Site O and Site S. Inorganic constituents were identified on a random basis throughout the five disposal sites.

4.6 Terrestrial Invertebrate Tissue Concentrations

In order to provide site-specific data regarding the potential exposure of COPECs that have been accumulated by terrestrial invertebrates to insectivorous or omnivorous wildlife consuming them, terrestrial invertebrate tissue was collected from the five disposal sites. One off-site and four composite onsite samples were collected. Actual sampling procedures are outlined in *Floodplain Area Field Sampling Report* (AMEC, 2003a). Terrestrial invertebrate tissue concentration results are summarized in Table III-A6.

As noted in AMEC (2003a), invertebrate specimens were collected either by sweep net or by hand. The total mass of terrestrial invertebrates required could not be collected at each sample location. While the collection success was variable based on available habitat (e.g., good habitat in Site Q - poor habitat in Sites R and S), the large amount of invertebrates needed could not be satisfied in the allotted sampling time. As such, invertebrate samples were composited within a single site, or in some instances, across several contiguous sites. Compositing was conducted only following concurrence with the USEPA oversight contactor. Following the compositing, there still remained some sampling areas that did not have sufficient biomass to conduct all of the required analyses. Therefore, analyses of the insect samples were prioritized in the following order: 1) dioxins/furans, 2) PCBs, 3) metals, 4) pesticides/herbicides, and 5) SVOCs. The identification of invertebrate samples is presented below.

TERRESTRIAL INVERTEBRATE SAMPLING MATRIX		
Sample Name	Composite of Sites:	Analyses
IN-Q1	Q-9, Q-10, Q-13, Q-17, Q-18, Q-19 and Q-20	dioxins/furans, SVOCs, metals, pesticides/herbicides, PCBs
IN-Q2	Q-11, Q-12, Q-14, Q-15 and Q-16	dioxins/furans, SVOCs, metals, pesticides/herbicides, PCBs
IN-OS1	Offsite areas OS-2, OS-3 and OS-4	dioxins/furans
IN-ROS1	O-1, O-2, O-3, R-1, R-2, R-3, R-4, R-5, S-1	dioxins/furans, metals, PCBs
IN-P1	P-1, P-2, P-3, P-4	dioxins/furans, metals, PCBs

The highest concentrations of dioxins/furans were identified in Q2 and ROS1. Pesticide and herbicides were also identified in those two samples. In general, SVOCs were not detected in invertebrate samples for which the analysis was run.

4.7 FOOD CHAIN MODELING EXPOSURE POINT CONCENTRATIONS

A generalized food chain model was used to estimate exposures of higher trophic level organisms to chemical concentrations in soil, surface water, sediment and prey items. Exposure to each COPEC was estimated by calculating an average daily dose (ADD) using (1) exposure media-specific concentrations, (2) estimated or measured exposure-point concentrations for prey/food, and (3) receptor-specific exposure parameters. The ADD represents the modeled amount of a chemical that an individual member of a receptor population could ingest if the individual foraged at least a portion of the time within the area used to develop the exposure-point concentrations. Inorganic and organic constituent concentrations used as input values into the food chain model included:

1. Surface soil data;
2. Surface water data from the Mississippi River and the pond;
3. Plant tissue residue data collected from the vicinity of the surface soil sampling locations;
4. Terrestrial invertebrate tissue residue data collected from the disposal sites in the vicinity of the surface soil sampling locations;
5. Earthworm tissue residue data developed as part of earthworm bioaccumulation tests; and
6. Fish tissue residue data obtained from fish collected in the pond and from fish collected in the Mississippi River by Menzie-Cura from prior investigations.

ADDs for ROIs (osprey, mink, prairie vole, short-tailed shrew and red fox) were developed based on procedures outlined in Sample *et al.* (1996) and USEPA (1993) and presented in the USEPA-approved Ecological Risk Assessment Work Plan. The total ADD is the sum of ADDs for each of the pathways (i.e., food, surface water, and sediment and/or soil), adjusted for the seasonal duration of exposure and normalized to body weight. The equations used for calculating the ADD are described in the Ecological Risk Assessment Work Plan (Appendix I-A).

In accordance with the USEPA-approved BERA work plan, the development of exposure estimates was made for each of the wildlife ROIs, except the red fox, on a site-by-site basis. For the red fox, the five Sites were considered in total in the determination of exposure estimates. Specific exposure parameters used in the development of the food chain models are presented in the USEPA-approved Ecological Risk Assessment Work Plan. Other assumptions used in the development of the food chain model are presented below:

1. Duplicate samples were treated as discrete samples.
2. For the osprey modeling procedure, 7 SVOCs and 1 herbicide were eliminated as no appropriate avian TRV could be developed.
3. For all food chain modeling receptors, calcium, iron, magnesium, potassium and sodium were eliminated from the calculations as they are considered essential nutrients.
4. For "Total PCB" calculations, if the individual homologs were all undetected (U) for a sample, then the highest detection limit was used and given an indicator of "U" as well. In samples with at least one detected homolog concentration, only the detected values were summed to give a total PCB value.
5. For the food chain modeling calculations, if there were at least 20 samples in a sample set, then an arithmetic average and 95% UCL value were calculated. If there were fewer than 20 samples, the arithmetic mean and maximum values were used.
6. There were some instances in which a 95% UCL could not be calculated for sample sets greater than 20. If all of the sample concentrations were equal to each other, the standard deviation was 0.0 and the UCL could not be generated. In those cases, the maximum value was substituted for the UCL.
7. For the fox food-chain modeling, it was assumed that the fox would eat equal amounts of shrews and voles so the dietary percentage of each was calculated by dividing the dietary percentage of mammals (68.8%) by two; = 34.4% voles; 34.4% shrews.
8. Surface water consumption was not used as an input parameter for the vole and shrew modeling as rodents obtain their water through ingestion of plants, rain from puddles, and dew (Vaughn, 1986)

9. 2-4 Dimethylphenol was not included in the modeling for the prairie vole for off-site areas, as well as Areas O, P, R and S because the plant tissue data for this chemical were rejected.

10. 2-4 Dimethylphenol was not included in the modeling for the short-tailed shrew or the red fox because the terrestrial invertebrate tissue data were rejected for this chemical. Because the shrew did not have a calculated average daily dose (ADD) for 2-4 Dimethylphenol, it could not be carried over to the fox model.

11. To calculate the concentration of a compound within the prairie vole for use in the fox modeling equations, the corresponding plant concentrations were multiplied by a plant-to-herbivorous mammal (deer mouse) bioconcentration factor (BCF). (Note: the BCF for 2,3,7,8-TCDD was used for the TEQ calculation). For those compounds lacking a BCF, the vole's average daily dose (ADD) was used instead.

12. The concentrations of compounds within the shrew for use in the fox modeling equations were equal to the shrew's average daily dose (ADD).

13. For the mink and fox modeling, the background locations (upstream for mink, off-site for fox) were given a percentage of ingested pond water (mink and fox) and/or pond fish (mink) of 0.0%. This is because the pond is considered an on-site hazard and it was not considered appropriate to include in the background modeling evaluations.

14. For the mink and the fox modeling, only the river surface water values closest to the shore (50 feet from the bank) were used in the calculations. The rationale was that the water closest to the rivers edge would be the only water theoretically accessible to these animals. For the mink and the fox modeling, as the surface water samples from the 50-foot transect line were not analyzed for dioxins/furans, samples from the 150-foot transect were used.

15. For the mink and osprey modeling, the river fish samples that were used as input parameters (Menzie-Cura, 2001) were not analyzed for metals. Fish tissue concentrations for metals were developed by multiplying the surface water metals data by a surface water-to-fish bioconcentration factor (BCF). If a BCF was not available for a particular metal, the average of the available inorganic BCFs was used as per USEPA, (1999a). This is a very conservative approach to estimating body burdens of metals in fish. In general, metals did not exceed their ambient water quality criteria, however with

certain metals, such as mercury, the calculated uptake of metals resulted in the estimation of a potential risk.

16. Individual dioxin/furan congener sample concentrations were multiplied by their appropriate TEFs for either fish (pond fish, pond surface water and pond sediment HQs), birds (osprey modeling) or mammals (surface soil HQs; vole, shrew, fox and mink modeling). In order to calculate TEQs, all dioxin/furan concentrations were multiplied by their TEF then added together to obtain the TEQ.

17. As the mink and osprey are wide ranging organisms that feed on prey that also is wide ranging (fish), estimated body burdens were calculated using average COPEC concentrations for input parameters. Likewise, red fox prey species estimated body burdens were calculated using average COPEC concentrations as this animal is also likely to forage over a large area. Additionally, the discontinuity of the disposal sites makes it highly unlikely that a red fox would travel from site to site to forage (*i.e.* Site P is separated from Site Q by industrial areas characterized by buildings, pavement and debris areas. Instead, the red fox would most likely move from disposal sites to neighboring undeveloped areas. As such, the average concentration is most reasonable for this organism.

Appendix III-B (Tables III-B1 through III-B13) summarizes the calculations and concentrations used in the food chain modeling.

As discussed in Sections 4.3 through 4.5 plant tissues and terrestrial invertebrates were collected from the Area 2 Sites for analysis of tissue body burdens to provide further input into the food chain modeling. Using soil collected from the Area 2 Sites, earthworm bioaccumulation tests were conducted and resulting tissue body burdens were determined through laboratory analysis of the worms Section 4.5. Data regarding the concentrations of inorganic and organic COPECs identified in the plant tissue was only used as a component of the food chain modeling for the prairie vole, which was carried through to the red fox modeling. Data regarding the concentrations of inorganic and organic COPECs identified in the earthworm tissue were only used as a component of the food chain modeling for the short-tailed shrew. Data regarding the concentrations of inorganic and organic COPECs identified in the invertebrate tissue were only used as a component of the food chain modeling for the short-tailed shrew. Food chain modeling information from the prairie vole and the short-tailed shrew was then fed into the food chain modeling for the red fox.

4.8 Exposure Point Concentration (EPC) Development

For the terrestrial ecosystem (surface soil, plant tissue, terrestrial invertebrate tissue and earthworm tissue) and aquatic ecosystem within the floodplain, a maximum and arithmetic average concentration was calculated for each analyte found at each one of the disposal sites. These values were utilized in the determination of risks to vermivorous wildlife (short-tailed shrew) and herbivorous wildlife (prairie vole). For the more wide-ranging species (mink, osprey, and red fox), average concentrations calculated from site-wide data were utilized for the risk calculations. The rationale is that it is very likely that species that forage over a wide area would be exposed to a range of concentrations and the average calculation is the most appropriate exposure scenario for these organisms. As per the USEPA-approved work plan, no initial screening of the analytes was conducted.

Section 5

5.0 AQUATIC ECOLOGICAL EFFECTS ASSESSMENT

The effects characterization is a qualitative and/or quantitative description of the relationship between the concentrations or dose of a COPEC and the nature of possible adverse effects elicited in exposed receptors, populations, and/or ecological communities. The results of the aquatic effects assessment and the aquatic exposure assessment (Section 3.0) are combined to characterize the risks to ROIs posed by COPECs in Section 7.0.

5.1 AQUATIC ECOTOXICOLOGICAL BENCHMARKS

Several of the measurement endpoints are associated with the comparison of Site-related data to ecotoxicological benchmarks for various media. The measurement endpoints are used in the evaluation of the potential for ecological risks to receptors in the Mississippi River adjoining the Sites. These benchmarks are risk-based screening concentrations that were used to evaluate the concentrations of chemicals detected in sediments and surface water in the Mississippi River. They are species- and chemical-specific values, and they typically represent chemical concentrations in a matrix below that adverse effects will not likely occur.

However, the benchmarks were not developed to serve as reference levels that will trigger specific remedial actions, if exceeded. The exceedance of a benchmark is not confirmation that an ecological impact is occurring. Rather, the benchmarks are primarily intended to help focus and prioritize project objectives and data requirements during the planning and implementing of site-specific investigations, by identifying constituents and particular areas of sites that may pose potential risks to aquatic and terrestrial ecological receptors.

5.1.1 SEDIMENT BENCHMARKS

For freshwater sediment screening, several benchmark values were available in the literature (Blanchet *et al.*, 2001). The benchmarks chosen for this report follow the hierarchical approach outlined in the Ecological Risk Assessment Work Plan (Section 12.0 of the SSP). These benchmarks are summarized in Table 5-1. The benchmarks that were used in the BERA are presented below.

- Ontario Lowest Effect Level (LEL) and Severe Effects Level (SEL): Concentrations of COPECs were first compared against Ontario LEL and SEL values prepared by the Ontario Ministry of the Environment (Persaud *et al.*, 1993). When both LEL and SEL values were available, constituent concentrations that exceeded the lower screening criteria (LEL) were retained as COPECs.

  The SEL values for non-polar organics (SVOCs, PCBs, pesticides) were normalized to TOC content of the sediment (Persaud *et al.*, 1993). The average TOC content of all sediment samples from the Site was calculated at 5922 mg/kg. SEL values for all nonpolar organic compounds were calculated by multiplying SEL values (Persaud *et al.*, 1993) by the fraction organic carbon (f_{oc} = 0.0059).

  No screening values were available for PCB homologs. Hence, individual homologs were added to calculate Sum-PCBs at each location as described in Section 3.1.8, and the maximum of Sum-PCBs was compared against LEL and SEL values for total PCBs.

- Threshold Effect Level (TEL) and Probable Effect Level (PEL): If Ontario Sediment Quality Guidelines (SQGs) were not available for a particular compound then TEL and PEL values were used. TELs are concentrations below which adverse effects are not expected to occur on sediment dwelling organisms, and PELs are concentrations above which adverse effects are expected to occur in sediment dwelling organisms frequently (MacDonald *et al.*, 2000).

  Of those compounds that did not have LEL/SEL values, only two compounds (bis(2-ethylhexyl)phthalate and 2,3,7,8-TCDD) had TEL/PEL values. The TEL/PEL values for bis(2-ethylhexyl)phthalate are from USEPA (1997c) and the TEL/PEL values for 2,3,7,8-TCDD (TEQ) are from Smith *et al.* (1996). TCDD TEQ concentration at the Site was calculated as described in Section 3.1.8. The TEL benchmarks were lower than PEL values, and constituent concentrations were screened against TELs.

- Other Benchmarks/Screening Values: None of the above benchmarks were available for the following compounds, and therefore, other screening numbers (not listed in the work plan) were calculated or were adapted from the literature as described below:

Benzo(b)fluoranthene: The toxicity of this compound was assumed to be the same as that of benzo(k)fluoranthene, and LEL and SEL values for benzo(k)fluoranthene were used.

Endrin Aldehyde, Endrin Ketone: No screening values were available for these two compounds; instead benchmarks (LEL/SEL) for Endrin were used for these two compounds on the assumption that the ecotoxicological effects and behavior of the three compounds are similar.

Pentachlorophenol: The Washington No Effects Level (Washington DEP, 2003) was used for screening.

Aluminum: USEPA ARCS (Assessment and Remediation of Contaminated Sediments) values (USEPA, 1996; Jones *et al.*, 1997) (Probable Effects Concentration (PEC) and No Effect Concentration (NEC)) were used.

Barium: The Apparent Effects Threshold (AET) value was used as a benchmark for barium.

Styrene, MCPP: For styrene and MCPP, benchmarks values were calculated using the equilibrium partitioning approach. The equilibrium partitioning approach is applicable to nonionic organic chemicals with log K_{ow} values from 2.0 to 5.5 (USEPA, 1997c). The sediment quality benchmark (SQB) was calculated using the following approach (Jones *et al.*, 1997):

$$SQB = f_{oc} \times K_{oc} \times WQB$$

Where,

f_{oc} is the site-specific fraction organic carbon (0.0059)

K_{oc} is the partition constant of the chemical to organic carbon, (values adopted from Risk Assessment Information System (RAIS) database)

WQB is the water quality benchmark

For styrene, a K_{oc} value of 517.8 L/Kg, and Illinois chronic aquatic toxicity criterion of 8 *ug/L* were used to derive a SQB of 611 *ug/kg*.

For MCPP, a K_{oc} value of 48.6 L/Kg, and Canadian Aquatic Life criterion (freshwater) value for phenoxy-herbicides of 4 *ug/L* (Canadian Environmental Quality Guidelines, 2002) were used to derive a SQB of 1.15 *ug/kg* under the assumption that the general value for phenoxy-herbicides is appropriate for MCPP.

Beryllium: The EqP concept was applied for beryllium by substituting solid-water partitioning constant (K_d) for K_{oc} ($K_d = K_{oc} \times f_{oc}$). A Tier II SCV of 5.09 *ug/L* (Suter and Tsao, 1996) was used as the water quality benchmark. Using a K_d value of 790 L/kg (RAIS Database) and a WQB of 5.09 *ug/L*, a SQB value of 4.02 mg/kg was calculated and was used for screening.

- Other Chemicals: For the following compounds, no screening values were available and SQBs could not be calculated using the EqP approach. These compounds were retained as COPECs.
  - *2,4-D* – This compound is ionic (pK_a = 2.87) and hence the EqP approach could not be used to calculate a SQB.
  - *Dalapon*: This compound has a low K_{ow} value (1.68), and hence the EqP approach was not used to calculate a SQB.
  - *p-Chloroaniline* – This compound is ionic (pK_a = 3.98) and hence the EqP approach could not be used to calculate a SQB.
  - *Vanadium* –No benchmarks are available.

5.1.2 Surface Water Benchmarks

For freshwater screening, the benchmarks chosen for this report follow the hierarchical approach outlined in the Ecological Risk Assessment Work Plan (Section 12.0 of the SSP). These benchmarks are summarized in Table 5-2. The benchmarks that were used in the BERA are presented below.

- National Ambient Water Quality Criteria (NAWQC): Surface water COPECs were first screened against acute and chronic National Ambient Water Quality Criteria (NAWQC) (USEPA, 2002). When both acute and chronic values were available, the chronic value (which is also the lower value) was used for screening.

  NAWQC for some metals (cadmium, chromium copper, lead, nickel, silver, and zinc) are functions of water hardness, and for these metals, acute and chronic criteria were calculated using the site-specific (Site-average) hardness value of 232 mg/L (as $CaCO_3$). For copper, zinc, lead, and mercury, cadmium, chromium, nickel, and silver the chronic and acute criteria for total metals were derived from dissolved fraction criteria using appropriate conversion factors specified in the NAWQC document (Table 5-3).

- Illinois Aquatic Life Criteria: If an NAWQC was not available for a chemical, then the surface water compound concentrations were screened against Illinois Aquatic Life Criteria (Illinois EPA, 2003. When both Acute Aquatic Toxicity Criteria (AATC) and Chronic Aquatic Toxicity Criteria (CATC) were available, the chronic value (which is also the lower value) was used for screening.

- Tier II Secondary Values: If NAWQC or Illinois Aquatic Life Criteria were not available for a chemical, then acute and chronic Tier II Secondary Values (SVs) assembled by Suter and Tsao (1996) were used for screening purposes. Chronic values were used for screening purposes.

- Lowest Chronic Value (LCV): If Federal (NAWQC), State (IL Aquatic Life Criteria), or Tier II SVs were not available, then the Lowest Chronic Value (LCV) for all organisms developed by the Oak Ridge National Laboratory (Suter and Tsao, 1996) was used. LCVs compiled by Suter and Tsao (1996) were used for calcium, magnesium, potassium and sodium.

- Other Criterion: None of the above numeric criteria was available for dichloroprop, an herbicide. For this compound, toxicity values for freshwater fish were searched for in the USEPA ECOTOX database (http://www.epa.gov/ecotox). Nine records (1 carp, 3 bluegills, 4 rainbow trout, and 1 brown trout) of fish toxicity (LC_{50}) were found. The geometric mean toxicity concentration (LC_{50}) was calculated as 6.4 mg/L. From the estimated LC_{50} value, a no-effects concentration (NOEC) was calculated by multiplying by a safety factor of 0.01. The resulting concentration of 64 *ug/L* was used as screening value for this compound (Note: as shown in Table 5-2, this derived value for dichloroprop is within the range of the acute and chronic benchmark values for 2,4-D, a structurally similar herbicide).

5.2 FISH TISSUE EFFECTS DETERMINATION

Potential for adverse effects on Site fish is evaluated by comparing concentrations of COPECs in fish tissue to body burden-based toxicity reference values (TRV) that have been reported to cause adverse effects in similar organisms. Maximum concentrations of COPECs in fish tissue from the Site were compared to No-Effect TRVs reported in the literature.

For the COPECs identified in fish tissue (Table 5-4), body burden-based TRVs were compiled primarily from two sources: the database compiled by USEPA (Jarvinen and Ankley, 1999), and the *Environmental Residue Effects Database (ERED)* (USACOE, 2002). These databases are compilations of data, taken from the literature, where biological effects and tissue contaminant concentrations were simultaneously measured in the same organism.

Both databases have information for various freshwater and saltwater species of fish and invertebrates. Since the Site has freshwater, only freshwater species were selected to develop the body burden-based TRVs. To the extent possible, only those fish species that are found in the Site area were selected. If none of the known site species were available, other closely related species found in streams and rivers were selected (for example, fathead minnows, which may not occur at the Site, might represent all warm water fish on a site – USEPA, 1994). Generally, body burden-based TRVs were available for several different life stages (e.g., adult, juvenile), exposure routes (e.g., ingestion, absorption), biological effects (e.g., growth, mortality), endpoints (e.g., LOED, NOED, LC_{50}), and fish tissues (e.g., whole body). When multiple data were available for a COPEC, data that closely matched the fish that were sampled and measurement endpoint (e.g., adult life stage, growth/reproduction/mortality effects, whole body analysis) were selected.

When data were available for multiple endpoints, the chronic No Observed Adverse Effect Level (NOAEL) and Lowest Observed Adverse Effect Level (LOAEL) were preferred over LD_{50} (Lethal Dose, 50% mortality) values. All body burden-based TRV benchmark values were converted to the equivalent of a NOAEL. If multiple endpoints were used, a LOAEL was converted to a NOAEL by multiplying by a safety factor of 0.1 and LD_{50} values were converted to NOAELs by multiplying by a safety factor 0.01 (factor of 0.1 for LD_{50} to LOAEL and another factor of 0.1 for LOAEL to NOAEL). Then the geometric mean of all the endpoints was calculated and used as the TRV for that COPEC. The data used for deriving the body burden-based TRV values are summarized in Appendix II-I.

For several compounds that did not have body burden-based TRVs, data for structurally similar compounds were used as surrogates: 3-methylphenol was used as a surrogate for 2-methylphenol (only trout fish data was available for 4-methylphenol and hence not used); beta-BHC was used as a surrogate for alpha-BHC; chlordane was used as a surrogate for gamma- and alpha-chlordane; endosulfan was used as a surrogate for endosulfan I; and endrin was used as a surrogate for endrin aldehyde.

2,4-Dichlorophenol is an ionic compound and its toxicity values are a function of the pH of the water. Toxicity data are available for experimental pH values of 6, 8, and 10. The average pH of surface water at the Site is 7.93 (Table 3-2) and therefore TRV data for pH 8 were used. For dioxin/furan compounds, TRVs were compiled for 2,3,7,8-TCDD (for comparison to TEQ) as well as for individual congeners.

For three herbicide compounds (2,4,5-T, 2,4,5-TP (Silvex), and MCPP), no TRV values were available. For these compounds, TRVs were derived from toxicity data available on USEPA ECOTOX database (http://www.epa.gov/ecotox). The ECOTOX database provides chemical toxicity information for aquatic and terrestrial life. For fish species that represent the habitat at the Site, concentrations of a particular chemical in test solutions (mg/L) that resulted in 50% mortality (LC_{50}) of test organisms after 24 to 96 hours of exposure were used. These toxicity values were converted to fish residue values by multiplying with an appropriate bioaccumulation factor (3.2 L/kg for all three compounds, taken from RAIS database). When multiple results were available, the geometric mean of the results was taken and multiplied with an appropriate uncertainty factor (0.01 to convert from LC_{50} to NOAEL as described earlier) to arrive at final body burden-based TRVs. The final body burden-based TRV benchmarks used for screening COPECs are summarized in Table 5-4.

Section 6

6.0 FLOODPLAIN ECOLOGICAL EFFECTS ASSESSMENT

As with the Aquatic Ecological Assessment, the determination of the possibility of adverse impacts occurring for several of the measurement endpoints was made by the comparison of Site-related data to ecotoxicological benchmarks for various media. These benchmarks were risk-based screening concentrations that were used to evaluate the concentrations of chemicals detected in surface soil, surface water and sediment in terrestrial/aquatic areas of interest at, and adjacent to, the Site. They are species- and chemical-specific values, and typically represent chemical concentrations in a matrix below which adverse effects will not likely occur.

This BERA utilized surface soil screening, surface water and sediment benchmarks identified in the *Ecological Risk Assessment Work Plan* (Section 12.0 of the SSP).

6.1 SURFACE SOIL PHYTOTOXICITY BENCHMARKS

To determine if the chemical concentrations in surface soils were toxic to the plant community, detected concentrations of COPECs in surface soil from each of the disposal areas were compared to phytotoxicity benchmarks identified in the literature. The soil benchmarks are based on data provided by toxicity studies in the field, or more commonly in greenhouse and growth chamber settings. These studies evaluated the effects of chemicals on various trees, wildflowers, grasses, and vegetable species.

A hierarchical approach to identifying soil benchmarks for screening potential constituents of concern in soils was used. The benchmarks in order of their evaluation are as follows:

- ONRL vegetative benchmarks (Efromyson *et al.*, 1997);
- USEPA Draft Ecological Soil Screening Levels (USEPA, 2000a);
- USEPA Region IV Recommended Ecological Screening Values for Soil (Friday, 1998);
- Canadian Soil Screening Values (British Columbia Regulation 375/96, 1997); and
- NOAEL/LOAEL identified in literature sources such as Eisler (2000).

6.2 Wildlife Benchmarks

Potential ecological impacts to wildlife ROIs were assessed by comparing exposures to ROIs to wildlife benchmarks identified from the literature. These screening benchmarks are identified as TRVs.

For each chemical and wildlife ROI, TRVs were derived to represent both a chronic no-observed-adverse-effect-level (NOAEL) and a chronic lowest-observed-adverse-effect-level (LOAEL). The NOAEL corresponds to the greatest exposure associated with no observed adverse effects on growth, reproduction, or survival. Exceedance of these values can be considered as potential impacts to sensitive individuals within the community. The LOAEL corresponds to the smallest exposure associated with observed adverse effects on growth, reproduction or survival. Exceedance of an LOAEL can be considered as a potential impact to the community as a whole. The TRVs utilized in this BERA are presented in Appendix III-C (Table III-C1)

6.3 Pond Surface Water and Sediment Benchmarks

The benchmarks used for screening of the pond sediment are defined in Section 5.1.1. The benchmarks used in the screening of the pond surface water are defined in Section 5.1.2. Because the ponds are ephemeral in nature, only the acute surface water and sediment values were used in evaluating these matrices. The use of chronic values implies that a long-term exposure may result to fish, amphibians, benthic organisms, or other aquatic receptors. However, the most significant long-term impact to the ponds is desiccation. Therefore, the BERA examined the potential for short-term or acute exposure to potential aquatic receptors.

6.4 COPECs in Pond Fish Tissue Effects Evaluation

Fish tissue collected from the large pond was analyzed and the data evaluated in the same manner as noted in Section 3.1.5 and Section 3.2.5. Effects determination of the risks posed by the presence of COPECs in fish tissue was made by a comparison to the same TRVs noted in Section 4.4. It is noted that the fish community present within the large pond in November 2003 no longer exists as the pond has dried up, killing the fish once present there.

Section 7

7.0 RISK CHARACTERIZATION

Risk characterization combines Site-related exposures and ecotoxicological effects benchmarks to estimate the potential for ecological risks. Procedurally, the risk characterization is performed for each measurement endpoint by (1) screening all measured COPECs against toxicological benchmarks and reference concentrations (2) where possible, estimating the potential effects of the COPECs identified at the Site using biological data; (3) where possible, evaluating the bioassay data; (4) logically integrating endpoints in a weight of evidence approach to characterize risks, and (5) listing and discussing the uncertainties in the assessment. The Ecological Risk Assessment Work Plan (Section 12.0 of the SSP) details the risk characterization procedures used in this BERA.

Concentrations of COPECs in exposure media (sediments, surface water or surface soil) or in calculated or identified body burdens were evaluated by the mathematical comparison of a constituent concentration to an ecotoxicological benchmark. Risk characterization of the potential for impacts to upper trophic level organisms was conducted in the same manner, with the ADD being divided by the TRV for the development of a hazard quotient (HQ). HQs are calculated using the following equation:

$$HQ = \frac{\text{Media-Specific Constituent Concentration or Total Estimate of Exposure}}{\text{Ecotoxicological Benchmark (in comparable units)}}$$

If the HQ is less than one, then it is concluded that the potential for impacts to ecological receptors is absent or minimal. If the HQ is equal to, or greater than one, then it is concluded that a *potential* for impacts to ecological receptors exists.

It is again emphasized that an HQ equal to, or greater than one is not confirmation that an ecological risk is occurring, only an indication of the potential for a risk and the need to examine other lines of evidence for confirmation. The magnitude of the exceedance over one may be related to the magnitude of the potential for impact. The level of conservatism in the benchmark, however, must be taken into consideration. Comparison of chemical data to strongly conservative benchmarks may be overprotective of the resource and exaggerate the potential for an ecological risk.

Aquatic and terrestrial hazard quotients were derived as follows:

- For the aquatic risk assessment, HQs were conservatively estimated using the maximum Site concentration (HQ_{max}) or the 95 percent upper confidence limit (UCL) of the arithmetic mean in a given medium, based on the Land H-statistic (EPA, 2002b). HQs for COPECs in sediments and surface water are summarized in Appendices II-J and II-K, respectively.

- As outlined in Section 4.5, HQs were calculated for vermivorous wildlife (short-tailed shrew) and herbivorous wildlife (prairie vole) using the maximum and average concentrations for each identified constituent detected in the surface soil of each of the disposal areas. For the more wide-ranging species (mink, osprey, and red fox), HQs were calculated using average concentrations calculated from site-wide data.

Although the Work Plan states that EPCs for wildlife receptors are best expressed as average COPEC concentrations in prey, surface water, and sediment or soil within the receptors' foraging area, a different approach was used for the aquatic assessment since it focuses more on benchmark comparison (as opposed to dose calculations) where an the upper estimate of the mean may be more appropriate for conservatism.

For the floodplain assessment, HQs were summed to give a Hazard Index (HI). As with the HQs, an HI in excess of one indicates the *potential* for an ecological risk. Exceedances at a large number of sampling points indicates the potential for more widespread, or community-level risks, as opposed to the exceedance at a single point which suggests potential impacts only to an individual assumed to reside at that one location for its entire life. Additionally, exceedances of chronic benchmarks, but not acute values, were indicative of potential ecological risks to sensitive individuals. The exceedance of an acute value over a large number of sampling points was considered to be indicative of the *potential* for community-wide ecological risks.

Some authors argue that HIs serve as an indicator of order-of-magnitude possibility for impacts to a receptor. As such, HIs between 1 and 10 are within the range of uncertainty or conservatism of the assessment and therefore indicate only a slight possibility of an ecological risk. HIs in excess of 10 indicate a stronger potential for an ecological risk (Barnthouse and

Suter, 1986 and Edmisten Watkin and Stelljes, 1993). Additionally, most NOAEL-based TRVs were developed using a 10-fold uncertainty factor on the LOAEL TRV. Therefore, values that are less than the LOAEL, but higher than the NOAEL, do not necessarily suggest an impact as a response (an effect) to the constituent has not been observed.

The following sections describe the various measurement endpoints used in the evaluation of exposure of fish to Site-related COPECs.

7.1 Risk Characterization of Aquatic Receptors in the Mississippi River

With the exception of isolated sampling areas that are discussed in further detail below, there were generally no ecological risks associated with sediments at the Site-related sampling areas. Minimal risks were noted with sediments in the sampling location just downstream of Site R. For surface water, the only significant ecological risks were also associated with the sampling locations found just downstream of Site R. It is noted that the ecological risks at that sampling area will be mitigated upon full implementation of the Site R interim remedy, which is designed to control the flow of contaminated groundwater into the river.

Using the initial screening process outlined in Section 3.2, preliminary COPECs were identified and carried through the full risk screening process. The following 19 analytes were retained as COPECs in sediments following the initial evaluation:

- Acetone
- Chlorobenzene
- 1,2-Dichlorobenzene
- p-Chloroaniline
- Dieldrin
- Endrin aldehyde
- Heptachlor epoxide
- 2,4-D
- Dalapon
- MCPP
- Arsenic
- Barium
- Cadmium
- Copper
- Lead
- Manganese

- Nickel
- Vanadium
- Zinc

The following analytes were retained as COPECs in surface water:

- p-Chloroaniline
- 2,4-D
- Aluminum (total)
- Barium (dissolved, total)
- Copper (total)
- Iron (total)
- Manganese (total)
- Vanadium (dissolved, total)

The following sections outline the potential ecological risks identified with the Mississippi River and its aquatic ecosystem as they relate to Assessment Endpoint #1.

7.1.1 Evaluation of Chemical Data from Sediments (Measurement Endpoint 1.A)

The ability of the benthic community to perform its role as a prey base for fish was evaluated by comparing maximum concentrations of COPECs in sediments to conservative sediment quality benchmarks as described in Section 5.1.1. An HQ was calculated for each of the sediment COPECs. The results are summarized in Table 7-1. The distribution of COPECs through the sampling areas is shown in Table 7-2. For the following constituents, benchmark values were not available and therefore HQs were not calculated:

- p-Chloroaniline
- 2,4-D
- Dalapon
- Vanadium

The following COPECs had an HQ≥1 when the maximum concentrations at the Site were compared to the sediment quality benchmarks:

- Acetone
- Chlorobenzene
- 1,2-Dichlorobenzene
- Dieldrin

- Endrin aldehyde
- Heptachlor epoxide
- MCPP
- Arsenic
- Barium
- Cadmium
- Copper
- Lead
- Manganese
- Nickel
- Zinc

For these contaminants, HQs were also calculated using the 95% UCL concentration. The analytes having $HQ_{95UCL} \geq 1$ included the follow:

- Acetone
- 1,2-Dichlorobenzene
- Dieldrin
- MCPP
- Barium

Some general patterns were noted in the distribution of the sediment COPECs. The highest levels of VOCs, SVOCs, pesticides and herbicides were present in the R3 sampling area (just downstream of Site R), with the highest positive results seen along transects located 50 feet and 150 feet from the river bank. With the implementation of the interim groundwater remedy at Site R, this discharge will be eliminated. While the inorganic COPECs were identified with maximum concentrations producing HQs greater than one, as none of the HQs were greater than 10, it is concluded that the concentrations are not likely to produce significant ecological impacts. These concentrations are also within the range of concentrations typical to area sediments. This is supported by the sediment toxicity test results discussed in Section 7.1.2.

The following is a summation by chemical classification of the distribution of COPECs identified in river sediments.

<u>VOCs</u>: Concentrations of two VOC compounds, acetone and chlorobenzene, exceeded applicable benchmark values. Acetone was detected at most Site areas and at *all* reference locations. All acetone data were qualified "J" (estimated value). Concentrations of acetone exceeded benchmark values at both the Reference area (HQ_{max} = 19) as well as at the Site

(HQ_{max} = 57). Note that acetone data was qualified by the data validators, and acetone is a common laboratory artifact.

Chlorobenzene was detected only in areas R3 and R5. Elevated concentrations of this compound were detected at a few locations in the R3 plot area, along two transects closest to the bank (50 feet and 150 feet from the bank).

SVOCs: Two SVOCs (1,2-dichlorobenzene and p-chloroaniline) were detected only at the R3 plot area and were found only at the 50 feet and 150 feet transects. Concentrations of 1,2-dichlorobenzene were only slightly above the benchmark value (HQ = 1.3).

An HQ could not be calculated for p-chloroaniline since a screening benchmark value was not available. However, the maximum surface water concentration of this compound was detected in the R3 plot area.

Pesticides: Three pesticides (dieldrin, endrin aldehyde, and heptachlor epoxide) were detected at concentrations that exceeded sediment quality benchmarks. Dieldrin exceeded the benchmark value (2.0 *ug/*kg) at only one location (R6AU). Endrin aldehyde exceeded the benchmark at only one location, R3BM. Heptachlor epoxide exceeded its benchmark only at the R3BM location.

Herbicides: Two herbicides (2,4-D and dalapon) did not have sediment quality benchmarks. MCPP was detected at two locations in the R3 plot area in the Site, and at two locations in the reference area. At both reference and Site locations, concentrations exceeded the sediment quality benchmark value (HQ_{max} in Reference sediments was 400; HQ_{max} in Site sediments was 2174). This suggests that a significant contribution of off-Site MCPP is moving through the river adjacent to the Area 2 Sites.

Metals: Maximum concentrations of eight metals (arsenic, barium, cadmium, copper, lead, manganese, nickel, zinc) exceeded benchmark values, but HQs were all less than 10 indicating the potential for impacts to benthic invertebrate communities was slight. HQs of three of these metals (barium, cadmium, manganese) were also $\geq$1 in reference area sediments, but less than in site-related areas. The maximum barium concentration was found in the R2 plot area.

Concentrations of vanadium, for which no screening values were available, appear to be derived from natural sources. The average "background" concentration for vanadium in freshwater sediments is 50 mg/Kg (NOAA, 2002), and all Site concentrations were less than this value. Concentrations of most metals are distributed without any obvious pattern. Concentrations of lead and zinc were elevated at R3 and some R4 sampling locations compared with upstream or downstream areas. Data regarding the metals concentrations in sediments in site-related sampling plots suggests that the identified metals concentrations are typical of Mississippi River sediments, based on a comparison to the reference plot and concentrations identified in Mississippi River sediments in other areas (Boyer, 1984).

7.1.2 Evaluation of Sediment Toxicity Testing (Measurement Endpoint 1.b)

The results of the sediment bioassays indicated that there were no significant toxic effects in any of the site-related sediment samples. The Dunnett's one-way ANOVA assessment of *H. azteca* survival demonstrated that there was no significant difference in mean survival when samples from Site areas R2, R3, R4, R5, and R6 were compared to their respective control samples. The one sample that exhibited a statistically significant different mean survival is R1BM1S, which is in the reference area. The mean survival for R1BM1S is 92% compared to the mean control survival for R1, which was 100%.

Similarly, the Dunnett's one-way ANOVA assessment of mean *C. tentans* growth demonstrated that none of the sediment samples collected from plots R1, R2, R3, R4, R5 of R6 exhibited mean growth that was significantly lower than the mean growth of the corresponding laboratory control samples at an alpha of 0.05.

A statistical summary of the sediment bioassay results is presented in Table 7-3.

7.1.3 Evaluation of Bioaccumulation Testing

While not included as a measurement endpoint, the bioaccumulation tests were used to qualitatively demonstrate the ability of COPECs in sediments to be accumulated by biological organisms (Appendix II-L). Concentrations and distribution of remaining constituents of concern are discussed below.

<u>SVOCs</u>: Three SVOCs were detected in a small number of clams exposed to Site sediments. indeno(1,2,3-cd)pyrene and pentachlorophenol were detected in just one clam sample (R5CM1S) and bis(2-ethylhexyl)phthalate was detected in three samples (detected at 8.1% frequency). All three SVOCs were detected in the associated sediments, but at concentrations below sediment quality benchmarks.

<u>Pesticides</u>: No pesticides were identified in clam tissue at concentrations higher in site-related sediments than in reference sediments.

<u>Herbicides</u>: Only dichloroprop concentrations were substantially higher (> 2X) in Site samples (highest in R3 plot area) than in reference samples.

<u>PCBs</u>: No PCB homologs were detected in any of the clams exposed to Site sediment.

<u>Dioxins</u>: A single dioxin congener (OCDD) was detected in clams exposed to Site sediments and reference area sediments. The maximum biota concentration (45 *ug* /kg) was reported for clams exposed to sediments from the R4AD location, although this congener was not detected in sediments at this location.

<u>Metals</u>: Maximum concentrations of three metals (chromium, lead, and nickel) in Site biota were significantly higher (> 2X) than the maximum concentrations in the reference biota. The maximum clam concentrations for all three of these metals were from sediments collected from the R4AM location, although the sediment concentrations at this location were similar to other locations.

In comparing the COPECs that were identified in the clams to the COPECs identified in the fish tissue work, only four organic COPECs (4,4'-DDE; 4,4'-DDT; 2,4,5-T; and MCPP) were found both in fish and in clams. Of these four COPECs, all of the concentrations identified in the clams were below the USACOE *ERED* values for those constituents. It is noted that metals were not analyzed in the fish collected by Menzie-Cura

7.1.4 EVALUATION OF BENTHIC INVERTEBRATE COMMUNITY ASSESSMENT

Exposure of COPECs in sediments to benthic invertebrates could result in reduced abundance, diversity, or biomass of benthic macroinvertebrates. This reduction in prey base could then affect the fish population, particularly the bottom feeders. Effects are evaluated by comparing the composition and abundance of benthic macroinvertebrates within the Site to data from the Reference area.

Benthic community diversity and abundance was evaluated at only two locations – one at the Site area (R6AD) and another at the Reference area (R1BD). In the Site area, a total of 66 organisms belonging to 8 different taxa were identified. In the Reference area, a total of 25 organisms belonging to 6 different taxa were identified.

In addition to the data collected in this study, results of the benthic study in the Krummrich report (Menzie-Cura, 2001) were also evaluated. In the Krummrich work, benthic samples were collected from several locations adjacent to the Site (in the area between R2 and R3) and reference areas. In the Krummrich study, evaluation of the benthic community was confounded by the high-energy environment (strong currents and coarse grain sediments) present at the Site. Menzie-Cura considered these results to be inconclusive. They found a sparse benthic community at the Site and upstream reference area. Because of the nature of the environment, it was hypothesized that the benthic community is not a significant component of the fish prey base, and plankton, drift, and periphyton were likely to be more important component of the prey base. Table 7-4 summarizes the results of the benthic evaluation.

7.1.5 EVALUATION OF CHEMICAL DATA IN SURFACE WATER (MEASUREMENT ENDPOINT 1.C)

Several COPECs were identified in the surface water analysis. The maximum concentrations of p-chloroaniline, aluminum, barium, iron, and vanadium exceeded water quality guidelines but the HQs were only slightly above 1. Concentrations of metals (aluminum, barium, iron, and vanadium) in Site samples were only slightly higher than the reference samples and did not appear to be Site-related. In addition, dissolved concentrations did not exceed water quality standards for most metals, and therefore no significant risk is expected from the presence of these metals in surface water to fish populations. The HQ_{max} for p-chloroaniline is estimated at

800, and this compound may pose some risk to fish. Concentration of this compound was highest at R3AU location, where surface water toxicity tests revealed severe toxicity to *C. dubia* than in control samples. Maximum concentrations of this compound and other COPECs were also detected in sediments from this location. The implementation of the groundwater interim remedy at Site R will resolve the presence of these COPECs by placing a barrier to groundwater discharge into the river. Surface water HQs are summarized in Table 7-5.

The following compounds had HQs $\geq$1.

- p-Chloroaniline
- 2,4-D
- Aluminum (total)
- Barium (dissolved, total)
- Copper (total)
- Iron (total)
- Manganese (total)
- Vanadium (dissolved, total)

The HQ for p-chloroaniline was the highest (HQ = 800); barium had HQs of 13.5 and 16.3 (dissolved and total barium, respectively). All other COPECs had HQs $\geq$ 1, but <10. The presence of p-chloroaniline and 2,4-D was limited to the R3, R4, and R5 plot areas. Maximum concentrations of these two constituents were detected at the R3AU site, located on the transect closest to the riverbank.

Concentrations of several metals were much higher in "total" analysis than in dissolved fraction, indicating that they were associated with suspended solids in the water column. Since the dissolved portion of metals is considered to be the more relevant fraction for toxicity evaluation, HQs that were calculated for total metals were compared with the dissolved fraction of the same metal. The HQ_{max} for total aluminum is 1.47, whereas the HQ_{max} for dissolved aluminum is much less than 1 (HQ_{max} = 0.02). Similarly, the HQ_{max} for total Iron was 1.4, as compared to an HQ_{max} for dissolved Iron of only 0.09. These compounds are not likely to be toxic to aquatic organisms because the dissolved concentrations do not exceed water quality standards. Copper was not detected in dissolved fraction and dissolved manganese was detected in only one sample (frequency of detection <5%) at a very low concentration. Thus, even though the HQ_{max} for copper (total) and manganese (total) were $\geq$1, they are not likely to be toxic to aquatic organisms because the dissolved concentrations do not exceed water quality standards.

For the above constituents of concern, HQs were also calculated based on 95% upper confidence limit (UCL) on the arithmetic mean. Concentrations of p-chloroanailine and 2,4-D appeared to be log-normally distributed and for these calculations, the 95% UCL values were calculated using the H-statistic (Appendix II-J). Concentrations of metals appeared to be normally distributed, and for these analytes the 95% UCL was calculated based on the t-statistic. Those values are also listed in Appendix II-J. The following COPECs had an HQ_{95UCL} ≥ 1.

- p-Chloroaniline
- 2,4-D
- Aluminum (total)
- Barium (dissolved, total)
- Iron (total)
- Vanadium (dissolved, total)

Concentrations of each of these COPECs at the Site and reference area locations are summarized in Table 7-3.

7.1.6 EVALUATION OF SURFACE WATER TOXICITY TESTING (MEASUREMENT ENDPOINT 1.C)

The results of the surface water bioassays indicated that there were significant toxic effects at only a limited number of sampling locations. As described in Section 4.2.1, the mean survival rates for *P. promelas* from all Site samples were not different from their respective laboratory control samples (Table 7-6). Mean growth rates were also similar between the Site and the control.

For the *C. dubia* survival analysis, five samples had mean 7-day survival rates lower than the control samples. The samples R3AU1W, R4AM1W, R4AD1W, R5AD1W, and R6AM1W had mean 7-day survival rates of 0% (SD=0%), 66.7% (SD=49%), 70% (SD=47%), 70% (SD=47%), and 70% (SD=47%), respectively. The mean 7-day survival rate for the pooled control samples is 96% (SD=19%). The R3AU1W sample was the only sample shown to be acutely toxic to *C. dubia* (i.e., 0% survival at 48-hours). The Tukey's pair wise ANOVA analysis of both acute and chronic *C. dubia* survival data showed that the mean survival of sample R3AU1W is significantly less than all other sampling locations. The toxicity observed in R6AM1W is not conclusive because the survival of the *C. dubia* was not affected in the duplicate sample from the same location (R6AM2W).

The effect on reproduction (i.e., neonate production) of *C. dubia* is described below by sampling area.

Site Area R2: For area R2, the Dunnett's ANOVA test showed that although some field samples had mean neonate production levels that were significantly different (*higher*) than their respective control sample, none had mean neonate production levels that were less than their respective control sample.

Site Area R3: For area R3, all samples from the transect nearest to the shore (R3AU1W, R3AM1W, and R3AD1W) and the two most upstream samples from the middle transect (R3BU1W and R3BM1W) were shown to have mean neonate production that was significantly lower than their respective control sample. The downstream sample from the middle transect (R3MD1W) and the sample from the furthest most transect (R3CM1W) did not have mean neonate production levels that differed significantly from their respective control samples. For both the R3A and R3B sampling transects, neonate production was lowest for the upstream samples and increased in downstream samples. However, neonate production rates for downstream (R3A and R3B) samples were still less than the control.

Site Area R4: For area R4, the Dunnett's ANOVA analysis showed that the mean neonate production of field sample R4AD1W was significantly lower than its respective control sample. All other R4 samples did not have mean neonate production levels that differed significantly from their respective control samples.

Site Area R5: For area R5, the Dunnett's ANOVA test showed that although some field samples had mean neonate production levels that were significantly different (*higher*) than their respective control samples, none had mean neonate production levels that were less than the control.

Site Area R6: For area R6, one sample, R6BM1W, was shown to have a mean neonate production level that was significantly less than its respective laboratory control sample. None of the COPECs (in sediment or surface water) were present at levels exceeding benchmarks at this location. All other R6 samples did not have mean neonate production levels that differed significantly from their respective control samples.

7.1.7 EVALUATION OF FISH COPEC BODY BURDENS (MEASUREMENT ENDPOINT 1.D)

Exposure of COPECs in diets and in surface water may result in accumulation in fish tissue, which can affect the survival, growth and reproduction of the fish. Exposure levels in fish were evaluated by comparing the concentrations of COPECs in fish tissue collected from the Site to the levels in fish from reference areas (Section 3.2.4). Potential for adverse effects to Site fish was evaluated by comparing concentrations of COPECs in fish tissue to body burden-based TRVs identified in Section 5.2. As previously mentioned, fish tissue results were obtained by Menzie-Cura as part of the Krummrich work (Menzie-Cura, 2001).

As described in Section 3.2.4, concentrations of several dioxin congeners were higher in the upstream (UDA) and/or downstream (DDA) reference area fish than in the Site (PDA) area fish. Concentrations of two compounds (2-methylphenol and 4,4'-DDE) were higher in downstream reference fish samples than in Site fish samples.

Maximum concentrations of constituents of concern in fish tissue from the Site (i.e., PDA in Krummrich report) and reference areas were compared to body burden-based TRV values as described in Section 5.2. HQs for all COPECs in Site fish and reference area fish were <1, except for MCPP from the Site (HQ_{max} = 1.5). As described in Section 5.2, a large uncertainty factor was incorporated into the development of TRV for MCPP. Uncertainties related to the use of the TRVs for MCPP and other COPECs are discussed in Section 8.

7.2 FLOODPLAIN RISK CHARACTERIZATION

The floodplain assessment evaluated several assessment endpoints. Those endpoints included the potential for adverse changes in the survival, reproduction, and growth of populations of two piscivorous wildlife species that utilize the Mississippi River in the vicinity of the Area 2 Sites (Assessment Endpoint #2). Other assessment endpoints include the evaluation of the potential for adverse changes in populations resulting from exposures to COPECs in prey of vermivorous wildlife (Assessment Endpoint #3), herbivorous wildlife (Assessment Endpoint #4) and carnivorous wildlife (Assessment Endpoint #5), all of which may be utilizing the six disposal sites. While hypothetical risks to vermivorous wildlife (short-tailed shrews) were calculated, because the habitat within the disposal sites does not appear to be suitable for the support of large populations of this organism, the importance of this animal was more as a transfer mechanism for COPECs to higher trophic level organisms that would feed on potential specimens (carnivorous wildlife – red fox).

Finally, the floodplain assessment evaluated the potential for adverse changes in the survival, reproduction, and growth of potential fish populations within the two ponds located in the southern end of Site Q resulting from exposures to COPECs in surface waters and sediments (Assessment Endpoint #6). The ponds are ephemeral in nature and additional data has been collected and is being evaluated respective to the ponds. As such the pond evaluation included in this BERA is worst-case and reflective of historic conditions and not reflective of current circumstances.

The following sections describe the potential risks to floodplain receptors associated with the Area 2 Sites from possible exposure of COPECs.

7.2.1 Evaluation of Site-Wide Potential Ecological Risks to Piscivorous Wildlife

This section discusses potential adverse ecological risks as they relate to Assessment Endpoint #2. The floodplain assessment identified that possible ecological impacts to mink feeding on fish from the river and the pond were slight. Additionally, the potential for adverse ecological impacts to the osprey was minimal. Individual COPECs included only those constituents that had positive detections in surface water and fish tissue and that exceeded upstream (background) HQs. Of the identified COPECs, none of the average daily exposures exceeded their respective LOAELs indicating that any potential risks would be possibly seen in sensitive individuals and not the population as a whole. Table 7-7 lists the COPECs that were developed for the mink and osprey from adjacent river areas and Table 7-8 lists the COPECs that were identified for the mink and osprey from the downstream areas. The identified COPECs based on mink food chain modeling include:

- Nitrobenzene (only as a COPEC in adjacent river areas, did not exceed LOAEL)
- MCPP (only as a COPEC in adjacent river areas, did not exceed LOAEL)
- PCBs (did not exceed LOAEL)
- Dioxins/furans (did not exceed LOAEL)
- Aluminum (did not exceed LOAEL)
- Antimony (did not exceed LOAEL)

Mink HQs and HIs from the upstream (sample plot R1) location are shown in Table 7-9. Mink HQs and HIs from the river adjacent to the Area 2 Sites (sampling plots R2, R3, R4, and R5) are shown in Table 7-10. Mink HQs and HIs from the downstream sampling area (sampling plot R6) are shown in Table 7-11.

The estimated ecological risks for the mink were driven by the consumption of fish from the large pond. The pond is an ephemeral water body that will support a fish community on a temporary basis, only if fish are washed into the ponds through overbank flooding of the Mississippi River. Fish were collected from the large pond and analyzed for the presence of COPECs. If those fish are removed from the modeling (a reflection of reality as the community no longer exists) then the only COPECs identified for the mink are MCPP and antimony. The risks associated with those COPECs are slight, as they do not exceed the LOAEL TRV.

HQs and HIs are presented for the osprey in Tables 7-12 through 7-14. The only COPEC that was identified for the osprey was mercury. As with the mink, the LOAEL-based TRV was not exceeded. The driving factor for the exceedance was the concentrations of mercury in fish tissue from the river. However, mercury concentrations in fish were estimated using surface water concentrations and BCFs, as the fish collected by Menzie-Cura were not analyzed for metals. This is very conservative, as the surface water AWQC for mercury was not exceeded. The mercury HQ for the osprey for the area adjacent to the Site was 9.4. For the downstream area, the HQ for mercury was 9.2.

HIs of one or more (indicating a potential for an ecological risk) were found in most categories of COPECs (SVOCs, herbicides, PCBs, dioxins/furans, and metals). In examining the upstream values presented in Table 7-15, it is apparent that a significant contribution of COPECs is generated upstream of the area adjacent to the site. Other than the SVOC HI (which was driven by nitrobenzene), there was little difference between the upstream, adjacent, and downstream HIs.

In summary, the potential for impacts to higher trophic level predators feeding primarily on aquatic receptors (exposed to COPECs in the Mississippi River) is considered to be minimal. The risks that do exist will be eliminated with the implementation of the interim groundwater remedy at Site R or have been eliminated by the loss of the fish community within the ponds located at the southern end of Site Q.

7.2.2 Evaluation of Potential Ecological Risks to Herbivorous and Vermivorous Wildlife

The following sections discuss the potential for adverse ecological impacts to occur as a result of the exposure of herbivorous and vermivorous wildlife to COPECs in surface soil at each of the five disposal sites. This evaluation addressed potential ecological risks related to assessment endpoints #3 and #4.

7.2.2.1 Site P Potential Ecological Risks

Two measurement endpoints (potential impacts to short-tailed shrews and potential impacts to prairie voles) were used to assess the potential for adverse ecological impacts to occur at Site P. Based on the type of habitat found at Site P, the most reasonable receptor to be potentially impacted by the presence of COPECs at this site is the prairie vole. Based on the food chain modeling conducted for the vole, no HQs in excess of one were identified for any constituent (using the average concentrations), indicating that none of the COPECs identified in the soil at Site P have the potential to adversely impact higher trophic level organisms. Using the maximum concentrations, only selenium (a natural constituent in soils) exceeded both the NOAEL and LOAEL TRV. These values were based on comparison to off-site HQs and HIs developed for the prairie vole, using average concentrations (Table 7-16) and maximum concentrations (Table 7-17) HQs and HIs are presented for the vole based on both average chemical concentrations and maximum chemical concentrations for Site P in Table 7-18 and Table 7-19.

In addition to the food chain evaluation of the prairie vole, an evaluation of the ability of the vegetative community to provide habitat for wildlife was performed as part of impact to herbivorous wildlife measurement endpoint. The rationale for the approach was that plants that were physiologically impacted by the presence of COPECs at a given site would not be able to provide appropriate habitat for wildlife within that disposal site. However, the phytotoxicity benchmarks are highly conservative, which is even noted by authors of the benchmarks, and often relies on hydroponic data that is not realistic in terms of site-specific conditions.

Based on the screening of surface soil concentrations of organic and inorganic constituents, the COPECs listed below were noted for Site P. The location of the maximum HQ for the COPEC and its value are also presented in the following list. Table 7-20 summarizes the HQs and HIs calculated based on the comparison of surface soil chemical data to impact to vegetation benchmarks.

For Site P, individual COPECs based on a potential impact to plants included:

- Anthracene (location P2, HQ of 2.3)
- Fluoranthene (location P2, HQ of 17)
- Arsenic (location P1, HQ of 2.6)
- Chromium (location P1, HQ of 19)
- Lead (location P2, HQ of 3.4)
- Nickel (location P1, HQ of 1.6)
- Selenium (location P1, HQ of 7.0)
- Thallium (location P1, HQ of 1.3)
- Vanadium (location P1, HQ of 22)

It is noted that the maximum exceedances of the plant benchmarks were located at sample locations W-P-1 and W-P-2. During the field studies in support of this BERA it was noted that some of the vegetative community around W-P-1 appeared sparse, however, that was considered to be the result of poor soil conditions as much of the soil in that area looked like cinders. There were no obvious indications of phytotoxic (*i.e.,* dead/dying plants, wilting or chlorosis) effects at either of the locations.

Only fluoranthene and vanadium had concentrations that resulted in an HQ in excess of 10. Plant uptake of PAHs is by active and passive transport through the roots and possibly through the transpiration stream (Polder *et al.*, 1995). PAHs are rapidly absorbed/adsorbed by the roots but little translocation occurs to the upper plant parts (Polder *et al.*, 1995). PAH-induced phytotoxic effects are rare (Polder *et al.*, 1995). In plants, pentavalent vanadium (V^{5+}) is a potential inhibitor of several enzymes but the tetravalent ion (V^{4+}) does not affect the same enzymes. There are no reports of vanadium toxicity under natural field conditions (Pendias and Pendias, 1992).

Vanadium is present in all mammals; however, tissue concentrations are very low (ATSDR, 1992f) due to low gastrointestinal absorption in animals (i.e., less than 0.1 to 2.6 % in rats). Absorption in rats is higher in young animals due to a greater non-selective permeability in the undeveloped intestinal barrier. Dermal absorption is minimal. Due to low bioavailability, biomagnification of vanadium is unlikely (ATSDR, 1992f).

A food chain evaluation of the potential for ecological risks to the short-tailed shrew based on earthworm bioaccumulation data and terrestrial invertebrate data was conducted. However, it is noted that none of the surface soil and plant samples used as input parameters to the food chain model were collected in habitat suitable to support the shrew. As such, the food chain results for the shrew are not realistic. HQs and HIs calculated for the shrew using both average chemical concentrations and maximum chemical concentrations for Site P are shown in Table 7-23 and Table 7-24. For Site P, the individual shrew based COPECs included:

- MCPA (maximum concentration NOAEL HQ 46, LOAEL 9.1)
- PCBs (maximum concentration only NOAEL HQ 2.8, did not exceed LOAEL TRV)
- Dioxins/furans (maximum concentration NOAEL HQ 34, LOAEL 3.4)
- Arsenic (maximum concentration NOAEL HQ 46, LOAEL 4.6)
- Cobalt (maximum concentration NOAEL HQ 32, LOAEL 3.2)
- Mercury (maximum concentration only NOAEL HQ 1.2, did not exceed LOAEL TRV)
- Selenium (maximum concentration NOAEL HQ 3.8, LOAEL 2.3)
- Thallium (maximum concentration NOAEL HQ 24, LOAEL 2.4)
- Vanadium (maximum concentration NOAEL HQ 23, LOAEL 2.3)

None of the constituents significantly exceeded their respective LOAEL values.

Using both average and maximum concentrations from Site P, HIs were calculated for comparison to off-site locations (Table 7-25). HIs were developed on a chemical class by chemical class basis. For the prairie vole, under both average and maximum concentrations, only dioxins/furans at Site P had an NOAEL HQ (1.2) significantly in excess of its background NOAEL HQ (0.30). For the shrew, all of Site-related average concentration NOAEL HIs and all of the maximum concentration NOAEL HIs exceeded their respective off-site HIs. However, only PCBs and dioxin/furans significantly exceeded their respective background HIs.

Table 7-26 summarizes all individual COPECs identified for the various Site P receptors.

7.2.2.2 Site R Potential Ecological Risks

Only minimal ecological risks were identified with this Site. Noting that a cap that extends several feet deep covers this Site, this is not unexpected. The majority of the vegetative COPECs (Table 7-20) were metals. HIs for both the shrew and the vole were not significantly different than those off-site HIs used as background. It is noted that Site R does not provide any habitat for a short-tailed shrew, and that the assessment of the vole and the plants is a more appropriate assessment for the site.

For Site R, the COPECs based on a comparison to phytotoxicity benchmarks included:

- Aluminum (location R3, HQ of 200)
- Chromium (location R1 and R3, HQ of 17)
- Iron (location R1 and R3, HQ of 90)
- Manganese (location R1, HQ of 2.4)
- Vanadium (location R3, HQ of 200)

The plant COPECs in Site R are all inorganic constituents that are common components of soil. Aluminum is naturally occurring, beneficial nutrient found in many plants at concentrations of ten to hundreds of mg/kg (Pendias and Pendias, 1992). Chromium is not taken up to any great extent by plants due to its low solubility and strong adsorption to soils (Allen, 2002). No adverse affects were seen in plants with concentrations up to 2140 mg/kg, which is much higher than the maximum concentration observed at Site R. Maximum tolerated doses for chromium oxides were 3000 ppm and 1000 ppm (Eisler, 2000). Phytotoxicity in sensitive species occurs around one to two ppm, which may inhibit seed germination and growth of roots and shoots (Eisler, 2000). Symptoms of phytotoxicity include wilting of tops, root injury, chlorotic new leaves and brownish-red leaves (Pendias and Pendias, 1992) and these symptoms were not observed during the field investigations.

Iron is an essential macronutrient for plants and animals. Essential nutrients are generally not encountered by ecological receptors at concentrations sufficient to cause toxicity. For example, iron is present in soils and sediments at varying quantities dependent on the geology of the area and the presence of other chemical components. Iron is not considered to be toxic except when present in the ferrous [Fe(II)] and ferric [Fe(III)] forms, which may be toxic at exceedingly high concentrations (USEPA, 1986).

HQs and HIs for the vole based on both average chemical concentrations and maximum chemical concentrations for Site R are shown in Table 7-27 and Table 7-28. For Site R, the vole based COPECs using the maximum concentrations, included:

- Cobalt (maximum concentration NOAEL HQ 1.1, did not exceed the LOAEL)
- Mercury (maximum concentration NOAEL HQ 4.3, did not exceed the LOAEL)

These ecological risks were considered to be slight as the LOAEL was not exceeded and the HQs were less than 10.

In animals, absorption of cobalt is dependent on the compound type, dose, animal weight and nutritional status. In rats, approximately 30% of cobalt (as cobalt chloride) was absorbed. Iron deficiency increases cobalt absorption (ATSDR, 1992c). Within biological systems, mercury is transformed into compounds with high toxicities that are mutagenic, teratogenic, and carcinogenic with embryocidal, cytochemical and histopathological effects. However, mercury toxicity in mammals is modified by the age, sex, sexual condition, and diet of the animal as well as the season and other variables. A body burden of less than 250 µg/kg for mammals and a daily dose of 32 µg/kg-bw for birds have been suggested (Eisler, 2000).

HQs and HIs for the shrew based on both average chemical concentrations and maximum chemical concentrations for Site R are shown in Table 7-29 and Table 7-30. For Site R, the shrew based COPECs, based on the maximum concentrations, included:

- Dichloroprop (maximum concentration NOAEL HQ 1.2, did not exceed LOAEL TRV)
- MCPP (maximum concentration NOAEL HQ 14, LOAEL HQ 4.8)
- Dioxins/furans (maximum concentration NOAEL HQ 10, LOAEL HQ 1.1)
- Cobalt (maximum concentration NOAEL HQ 17, LOAEL HQ 1.7)
- Vanadium (maximum concentration NOAEL HQ 8.0, did not exceed LOAEL TRV)

Again, because the HQs were generally less than 10 and in two instances, the LOAEL was not exceeded, the ecological risks associated with the identified COPECs were considered to be slight for the short-tailed shrew.

Table 7-31 summarizes all COPECs identified for the various Site R receptors.

7.2.2.3 Site O Potential Ecological Risks

The assessment of Site O was based on the use of three data points for surface soil concentrations, one of which was located in sub-area Site O (North), and the earthworm bioaccumulation evaluation. As with the preceding two disposal sites, based on the type of habitat found in the site, the prairie vole is the most appropriate terrestrial receptor that was evaluated in this BERA.

For Site O, the COPECs that were identified by screening surface soil chemical data against phytotoxicity benchmarks included the following constituents (Table 7-20):

- Chlorobenzene (location O1, HQ of 5.8)
- Ethylbenzene (location O1, HQ of 4.4)
- Total xylenes (location O1, HQ of 16)
- Dieldrin (location O3, HQ of 360)
- Lindane (location O3, HQ of 640)
- PCBs (location O1, HQ of 18)
- Dioxins/furans (location O3, HQ of 2.3)
- Aluminum (location O3, HQ of 170)
- Arsenic (location O1, HQ of 1.1)
- Cadmium (location O1, HQ of 4.3)
- Chromium (location O1, HQ of 16)
- Copper (location O1, HQ of 2.7)
- Mercury (location O1, HQ of 140)
- Silver (location O1, HQ of 1.5)
- Vanadium (location O3, HQ of 14)
- Zinc (location O1, HQ of 19)

While a number of constituents were identified as exceeding these conservative benchmarks, there were no signs of phytotoxic stress in the vegetative community. COPECs that significantly exceeded their respective benchmarks included dieldrin, lindane, aluminum, and mercury.

The food chain modeling of constituents identified in soil and plants to a typical site related herbivore identified a few COPECs. HQs and HIs for the vole based on both average chemical concentrations and maximum chemical concentrations for Site O are shown in Table 7-32 and Table 7-33. The individual COPECs in Site O that posed a potential adverse impact to prairie voles included:

- PCBs (maximum concentration NOAEL HQ 11, LOAEL HQ 1.1)

- Dioxins/furans (maximum concentration NOAEL HQ 80, LOAEL HQ 8.0)
- Lead (maximum concentration only NOAEL HQ 1.7, did not exceed LOAEL TRV)
- Mercury (maximum concentration NOAEL HQ 3.7, did not exceed LOAEL)
- Thallium (maximum concentration only NOAEL HQ 3.9, did not exceed LOAEL TRV))

Lead had an HQ of 1.7, mercury had an HQ of 3.7 and thallium had an HQ of 3.9, indicating that the ecological risks associated with those constituents were minimal. As the HQs were generally less than 10 and in three instances, the LOAEL was not exceeded, the ecological risks to the prairie vole associated with the identified COPECs were considered to be slight.

Again, Site O did not have habitat that was advantageous to the short-tailed shrew. HQs and HIs for the shrew based on both average chemical concentrations and maximum chemical concentrations for Site O are shown in Table 7-34 and Table 7-35:
For Site O, the shrew based COPECs included:

- Heptachlor epoxide (maximum concentration NOAEL HQ 2.2, did not exceed the LOAEL)
- Pentachlorophenol (maximum concentration only NOAEL HQ 1.4, did not exceed the LOAEL)
- MCPP (maximum concentration NOAEL HQ 13, LOAEL HQ 4.4)
- PCBs (maximum concentration NOAEL HQ 310, LOAEL HQ 31)
- Dioxins/furans (maximum concentration NOAEL HQ 2,300, LOAEL HQ 230)
- Aluminum (maximum concentration NOAEL HQ 460, LOAEL HQ 46)
- Antimony (maximum concentration NOAEL HQ 5.1, did not exceed the LOAEL)
- Arsenic (maximum concentration NOAEL HQ 23, LOAEL HQ 2.3)
- Barium (maximum concentration only NOAEL HQ 3.0, did not exceed the LOAEL)
- Cadmium (maximum concentration only NOAEL HQ 1.1, did not exceed the LOAEL)
- Mercury (maximum concentration NOAEL HQ 51, LOAEL HQ 10)
- Selenium (maximum concentration only NOAEL HQ 1.2, did not exceed the LOAEL)
- Silver (average concentration only NOAEL HQ 2.4, did not exceed the LOAEL)
- Thallium (maximum concentration NOAEL HQ 24, LOAEL HQ 2.4)
- Vanadium (average concentration only NOAEL HQ 6.7, did not exceed the LOAEL)

PCBs and dioxin/furans had significantly higher HQs in comparison to the off-site areas. Site O (North) was the only area evaluated in this BERA where petroleum related constituents (chlorobenzene, ethylbenzene, and xylenes) produced elevated HQs for a terrestrial receptor.

Table 7-31 summarizes all COPECs identified for the various Site O receptors.

7.2.2.4 Site S Potential Ecological Risks

The assessment of Site S was based on the evaluation of a single data point for surface soil concentrations, plant and insect tissue concentrations and the earthworm bioaccumulation results. As with the preceding disposal sites, based on the type of habitat found in the site, the prairie vole is considered to be the most appropriate terrestrial receptor for this disposal area.

Site S had a number of constituents in surface soil that exceeded the impact to vegetation benchmarks. However, there were no obvious indications of phytotoxic effects. The vegetation at Site S showed excellent growth, and there were no indications of dead plants, wilting, or chlorosis. For Site S, the COPECs that were identified through the comparison of constituents in surface soil to phytotoxicity benchmarks included:

- 1,2,4-Trichlorobenzene (location S1, HQ of 18)
- 1,2-Dichlorobenzene (location S1, HQ of 37)
- 1,3-Dichlorobenzene (location S1, HQ of 1.0)
- 1,4-Dichlorobenzene (location S1, HQ of 7.5)
- 2,4,6-Trichlorophenol (location S1, HQ of 16)
- 2,4-Dichlorphenol (location S1, HQ of 4.6)
- Benzo(a)anthracene (location S1, HQ of 8)
- Benzo(a)pyrene (location S1, HQ of 5.4)
- Benzo(b)fluoranthene (location S1, HQ of 6.6)
- Dibenzo(a,h)anthracene (location S1, HQ of 1.8)
- Fluoranthene (location S1, HQ of 4.2)
- Indeno(1,2,3-cd)pyrene (location S1, HQ of 1.3)
- Naphthalene (location S1, HQ of 3.5)
- Pentachlorophenol (location S1, HQ of 15)
- Phenanthrene (location S1, HQ of 1.8)
- Pyrene (location S1, HQ of 2.8)
- 4,4'-DDT (location S1, HQ of 1.6)
- Beta-BHC (location S1, HQ of 26,000)
- Endrin (location S1, HQ of 10,000)
- Lindane (location S1, HQ of 150,000)
- PCBs (location S1, HQ of 25)
- Chromium (location S1, HQ of 21)
- Manganese (location S1, HQ of 1.3)

The pesticides in this sample (Beta-BHC, endrin, and lindane) had the highest HQs noted for the entire Sauget Area 2 Sites.

While a number of COPECs were identified that could theoretically impact plants, the food chain modeling that further evaluated the constituents identified in surface soil and plant tissue identified only a few COPECs. HQs and HIs for the vole based on both average chemical concentrations and maximum chemical concentrations for Site S are shown in Table 7-37 and Table 7-38. The individual COPECs in Site S that posed a potential adverse impact to prairie voles included:

- Pentachlorophenol (maximum concentration NOAEL HQ 2.9, did not exceed LOAEL)
- PCBs (maximum concentration NOAEL HQ 16, LOAEL HQ 1.6)
- Mercury (maximum concentration NOAEL HQ 2.2, did not exceed LOAEL)

PCBs were the most significant COPEC out of this group, though the LOAEL was only slightly exceeded.

HQs and HIs for the shrew based on both average chemical concentrations and maximum chemical concentrations for Site S are shown in Table 7-39 and Table 7-40. For Site S, the shrew based COPECs included:

- Chrysene (maximum concentration NOAEL HQ 1.7, did not exceed LOAEL)
- Pentachlorophenol (maximum concentration NOAEL HQ 65, LOAEL HQ 6.5)
- MCPA (maximum concentration NOAEL HQ 7.1, LOAEL HQ 2.4)
- PCBs (maximum concentration NOAEL HQ 360, LOAEL HQ 36)
- Cobalt (maximum concentration NOAEL HQ 14, LOAEL HQ 1.4)

The most significant COPEC based on the shrew modeling was PCBs.

Table 7-41 summarizes all COPECs identified for the various Site S receptors.

7.2.2.5 Site Q (South) Potential Ecological Risks

As with the preceding disposal sites, based on the type of habitat found in the site, the prairie vole is considered to be the most appropriate terrestrial receptor for Site Q (South) (Ballenger, 2000). Site Q (South) had a number of constituents in surface soil that exceeded the impact to vegetation benchmarks. However, there were no obvious indications of phytotoxic effects. The vegetation at Site Q (South) showed excellent growth, and there were no indications of dead plants, wilting, or chlorosis. For Site Q (South), the COPECs identified through a comparison of surface soil concentrations to phytotoxicity benchmarks included:

- Anthracene (location Q9, HQ of 3.4)
- Benzo(a)anthracene (location Q10 Dupe, HQ of 4.5)
- Benzo(a)pyrene (location Q10 Dupe, HQ of 5.3)
- Benzo(b)fluoranthene (location Q10 Dupe, HQ of 4.8)
- Benzo(k)fluoranthene (location Q10 Dupe, HQ of 4.4)
- Fluoranthene (location Q10 Dupe, HQ of 80)
- Pentachlorophenol (location Q11 Dupe, HQ of 1.2)
- Phenanthrene (location Q10 Dupe, HQ of 1.3)
- Alpha-BHC (location Q11, HQ of 80)
- Beta-BHC (location Q11, HQ of 820)
- Dieldrin (location Q11 Dupe, HQ of 700)
- Endrin (location Q9, HQ of 170)
- Dioxins/furans (locations Q11, and Q11 Dupe HQs < 10)
- Aluminum (location Q9 and Q11, HQ of 220)
- Antimony (location Q9, HQ of 9.4)
- Arsenic (location Q9, HQ of 3.3)
- Barium (location Q11 Dupe, HQ of 3.0)
- Cadmium (location Q11, HQ of 7.5)
- Chromium (location Q9, HQ of 660)
- Cobalt (location Q9, HQ of 1)
- Copper (location Q11 Dupe, HQ of 8.7)
- Iron (location Q9, HQ of 450)
- Lead (location Q10 Dupe, HQ of 9.8)
- Manganese (location Q9, HQ of 4.2)
- Mercury (location Q10 Dupe, HQ of 9.0)
- Nickel (location Q9, HQ of 17)
- Selenium (location Q11, HQ of 2.5)
- Silver (location Q11, HQ of 8.5)
- Thallium (location Q13, HQ of 1.9)
- Vanadium (location Q10 Dupe, HQ of 1.6)

The COPECs with significant exceedances of the benchmark included fluoranthene, alpha-BHC, beta-BHC, dieldrin, endrin, aluminum, chromium and iron. It is noted that out of the above 26 COPECs, 18 of the COPECs had HQs between 1 and 10, indicating that the ecological risks associated with exposure to plants is only slight. Most of the maximum exceedances of the impact to plant benchmarks were located in the center section of the southern area (Q-9, Q-10, and Q-11). Vegetation in this area was particularly thick. While Site Q (South) had one of the largest lists of constituents that exceeded plant benchmarks (though the majority of that number did not exceed an HQ of 10), the field studies conducted in support of the BERA did not find any indications of an impacted plant community anywhere on Site Q.

A number of COPECs were identified that could theoretically impact plants. However, the food chain modeling that evaluated potential ecological risks to an herbivore typically found at the site identified only a few COPECs. HQs and HIs for the prairie vole for Site Q (South) are shown in Table 7-42 and Table 7-43. For Site Q (South), COPECs for the prairie vole included:

- Dioxin/furans (maximum concentration NOAEL HQ 13, LOAEL HQ 1.3)
- Aluminum (maximum concentration NOAEL HQ 25, LOAEL HQ 2.5)
- Antimony (maximum concentration NOAEL HQ 1.9, did not exceed the LOAEL)
- Arsenic (maximum concentration NOAEL HQ 1.2, did not exceed the LOAEL)
- Cobalt (maximum concentration NOAEL HQ 1.2, did not exceed the LOAEL)
- Mercury (maximum concentration NOAEL HQ 2.4, did not exceed the LOAEL)

None of the COPECs identified above have significant exeedances of their respective TRVs. The potential for adverse ecological impacts to the prairie vole from these COPECs is considered to be slight.

HQs and HIs for the shrew based on both average chemical concentrations and maximum chemical concentrations for Site S are shown in Table 7-44 and Table 7-45. For Site Q (South), the shrew based COPECs included:

- Heptachlor epoxide (maximum concentration HQ 3.8, did not exceed LOAEL)
- PCBs (maximum concentration NOAEL HQ 36, LOAEL HQ 3.6)
- Dioxins/furans (maximum concentration NOAEL HQ 330, LOAEL HQ 33)
- Aluminum (maximum concentration NOAEL HQ 810, LOAEL HQ 81)
- Antimony (maximum concentration NOAEL HQ 28, LOAEL HQ 2.8)
- Barium (maximum concentration NOAEL HQ 12, LOAEL HQ 3.0)
- Chromium (maximum concentration NOAEL HQ 7.3, LOAEL HQ 1.8)
- Cobalt (maximum concentration NOAEL HQ 23, LOAEL HQ 2.3)
- Copper (maximum concentration NOAEL HQ 6.4, LOAEL HQ 4.8)
- Lead (maximum concentration NOAEL HQ 14, LOAEL HQ 1.4)
- Mercury (maximum concentration NOAEL HQ 5.6, LOAEL HQ 1.1)
- Selenium (maximum concentration NOAEL HQ 1.4, did not exceed LOAEL)
- Silver (maximum concentration NOAEL HQ 7.6, did not exceed LOAEL)
- Thallium (maximum concentration NOAEL HQ 33, LOAEL HQ 3.3)
- Vanadium (maximum concentration NOAEL HQ 10, LOAEL HQ 1.1)

Of those constituents, only dioxins/furans and aluminum had significantly elevated HQs in comparison to the off-site areas. As with the plant COPECs, the shrew COPECs (which are driven by the earthworm bioaccumulation tissue results) were most localized in the central section of Site Q (South)(sample locations Q-11, Q-12, Q-13 and Q-14).

Table 7-46 summarizes all COPECs identified for the various Site Q (South) receptors.

7.2.3 Evaluation of Site-Wide Potential Ecological Risks to Carnivorous Wildlife

A food chain evaluation of the potential for ecological risks to the red fox based on modeled consumption of shrews and voles was employed to assess whether COPECs at the individual disposal sites had the potential to impact upper trophic level organisms (Assessment Endpoint #5). The assumption was that red fox would feed on an equivalent amount of shrews and voles as part of foraging across the disposal sites. Therefore, body burden calculations for the shrew and vole served as a pathway from COPECs identified in surface soil to this upper trophic level organism. In keeping with OSWER Directive 9285.7-28P (USEPA, 1999c), the more critical receptor is the red fox.

Potential risks were calculated based on exceedance of TRVs and exceedance of HQs and HIs calculated for the red fox using off-site surface soil concentrations (Table 7-47). HQs and HIs are presented for the red fox using average chemical concentrations in Table 7-48.

Using the average input values of soil, surface water, plants and modeled prey body burdens (prairie vole and short-tailed shrews), COPECs for the red fox include:

- PCBs (NOAEL HQ 11, LOAEL 1.1)
- Dioxins/furans (NOAEL HQ 44, LOAEL 4.4)
- Aluminum (NOAEL HQ 130, LOAEL 13)
- Antimony (NOAEL HQ 130, did not exceed the LOAEL)
- Arsenic (NOAEL HQ 2.0, did not exceed the LOAEL)
- Cobalt (NOAEL HQ 4.4, did not exceed the LOAEL)
- Mercury (NOAEL HQ 2.8, LOAEL HQ 1.7)
- Thallium (NOAEL HQ 7.0, did not exceed the LOAEL)
- Vanadium (NOAEL HQ 2.3, did not exceed the LOAEL)

The most significant COPEC identified above, based on the exceedance of the TRVs in relationship to the off-site areas, is aluminum. As the red fox has a mean home range of 1,727 acres, it is highly unlikely that the disposal sites (total area less than 150 acres) would support a large population of red fox. Noting the discontinuity of the sites, it is more likely that a small number of fox utilize a portion of different disposal areas for foraging, moving between contaminated and non-contaminated areas. Additionally, the fence surrounding Site R would limit access of the fox to this disposal area. Site-wide COPECs, based on the red fox, are listed in Table 7-49.

7.2.4 Evaluation of Potential Ecological Risks Associated with Site Q (Ponds)

Risks to the fish populations within the two ponds located at the southern end of Site Q were evaluated using the same measurement endpoints as used to evaluate the risks to the fish population in the Mississippi River (Section 5.0). It is again noted that the evaluation of these measurement endpoints is for conditions within the ponds that no longer exist. At the time of the evaluation, the smaller pond was dry and the larger pond was significantly reduced in size and dried up shortly following sample collection. Both ponds have subsequently refilled; however, because overbank flooding has not occurred, a fish population within the ponds has not been reestablished. Recent data collected in June 2003 regarding surface water and sediment quality within both ponds will be evaluated and presented in an addendum to this BERA at a future date.

Sediment HQs were calculated through the comparison of COPECs detected in a single sediment sample (with a duplicate) collected in the large pond to acute conservative sediment benchmarks. As previously noted, because the ponds are ephemeral in nature, sediments and surface water COPECs were compared to acute benchmarks. Only dioxins and furans showed HQs in excess of one. However, both total TCDD (HQ of 4.0) and total TCDF (HQ of 2.0) had an HQ of less than 10, indicating only a slight possibility for an adverse ecological risk. Table 7-50 presents the HQs for the sediment sample from the large pond.

Table 7-51 presents the HQs calculated for the two surface water samples collected within the large pond based on a comparison of COPECs identified in the surface water to benchmarks. Of the analytes detected in P-11W, only aluminum (HQ of 10) and barium (HQ of 1.8) had HQs of greater than one. The potential for adverse ecological risks to be associated with these constituents is considered to be slight. Sample P-12W, which had metals analyzed on a filtered basis, did not have any detected concentrations of inorganic constituents exceed an acute benchmark.

Toxicity tests for both surface water and sediments did not indicate any significant toxicity to the test species.

Table 7-52 lists the HQs calculated through the comparison of pond fish COPEC tissue body burdens to body burden-based TRVs. Exceedance of these TRVs suggests that the COPECs within the fish tissues have the potential to adversely impact the fish. However, it is noted that the fishery community within the pond has been lost as the pond dried up. Noting that the fish in the pond were transplanted by flooding from the Mississippi River, it is most likely that the largest source of the COPECs carried by the pond fish originated within the Mississippi River. Table 7-53 lists the limited COPECs identified with the large pond.

7.3 ECOTOXICOLOGY

As previously outlined, a variety of COPECs have been identified for the various sites. However, ecotoxicological properties of the COPECs can mitigate the actual expression of predicted ecological risks. A receptor must be exposed such that the toxicant must first enter the body through an exposure route (inhalation, ingestion or dermal contact) before interacting with cells. This is a concept known as bioavailability. For the purpose of the BERA, the COPECs were assumed to be 100 percent bioavailable.

Bioavailability is an important concept in risk assessments because toxicological effects are not elicited merely from the presence of a chemical. Bioavailability is defined as that portion of the exposure concentration that actually reaches the systemic (arterial) circulation for distribution through the body and/or to the target organ. A chemical's bioavailability is affected by physical, chemical, and biological factors such as physical adsorption (*e.g.,* K_{oc}), lipid (fat) solubility and chemical adsorption, among others (Hamelink, *et al.*, 1994). Bioavailability determines if a chemical will bioaccumulate and/or biomagnify. Bioaccumulation is defined as the net accumulation of a substance in a living organisms from all routes of exposure. Biomagnification is the tendency of a chemical to accumulate to higher concentrations at higher trophic levels through dietary accumulation (Suter, 1993).

Appendix III-E discusses ecotoxicological expression and bioavailability of the various COPECs identified as part of the Floodplain Risk Assessment.

Section 8

8.0 UNCERTAINTIES IN THE RISK ASSESSMENT

There are several sources of uncertainty associated with ecological risk estimates. This includes initial selection of COPECs based on the sampling data, estimates of toxicity to ecological receptors based on limited laboratory data (usually on other species), and uncertainties in exposure and effects assessment. This section describes some of these uncertainties along with what actions have been taken to manage this uncertainty within the assessment.

8.1 COMPONENTS OF THE UNCERTAINTY ANALYSIS

Uncertainty in risk estimation has both qualitative and quantitative components. Qualitative uncertainty analyses are recommended by guidance (*e.g.,* USEPA, 1988c, page 96) and contribute to the confidence with which risk assessment conclusions can be drawn and applied *(USEPA 1989a; 1992a). Where possible, quantitative uncertainty analyses provide objective* measures of the relative confidence in conclusions and applications.

Uncertainty surrounding risk assessment conclusions has important implications for risk management (USEPA 1988c; 1998). However, uncertainty is not a single, generally applicable parameter. Uncertainty surrounding a risk estimate or application has a number of components, including parameter variability, calculation error and simplification, and the underlying reality of exposure assumptions and pathways (USEPA 1988c). It is important to understand that uncertainty includes both real variation (reflecting actual, mechanistic biological response ranges and variability in ecosystem conditions) and error (USEPA, 1997a). Thus, because biological systems are inherently uncertain and variable, some component of variability in risk estimation is due to a realistic reflection of ecological conditions, while another component is due to error or uncertainty introduced by the overall analytical process. Error is the component to be minimized, because this encompasses undesirable uncertainty that has been introduced by the assessment process. However, it is critically important to understand ecosystem variability because this represents an important component of the ecosystem within which risk management decisions must be made. Substantial differences exist between observations and conclusions made at the individual, population, and community levels of biological organization. For example, effects not manifested at the population or community levels (*e.g.,* mortality of only a few individuals) may not be observable with the type of studies implemented. The ramifications of this also include an understanding that, because the assessment level endpoints are protective of populations and communities and not individuals, the projected loss of a few individuals may not cause impacts that are important at the levels of assessment where risk management decisions are made.

Due to the multiplicity of potential receptor species and general lack of knowledge regarding their life cycles, feeding habits, nutritional requirements (*e.g.,* essential elements such as arsenic, trivalent chromium, selenium, zinc), and relative toxicological sensitivity, the uncertainty surrounding estimates of ecological risk may be substantially greater than those associated with human health risk assessment. The generic screening and regulatory criteria and TRVs used in this assessment are intended to provide conservative benchmarks, but it is important to note that no one approach to criterion/TRV derivation is adequate for all sites and all chemicals. The criteria/TRVs used in this assessment are all chemical-specific and as such cannot address the additive, antagonistic, or synergistic effects of the chemical mixtures typically found in the environment (Swartz *et al.,* 1988). Further, these criteria/TRVs do not take into account the structure and dynamics of the ecosystem present at the site, site-specific conditions regulating chemical contact and bioavailability, the potential toxicity of other constituents that were not quantified, or the pervasive influence of physical stressors associated with the disruption by human activities that is characteristic of an area that has been actively industrial for over 100 years, or the result of periodic flooding from the Mississippi River.

The evaluations presented herein were performed within a range of conditions defined by characteristics of the environment at the time field data were gathered. As such, data obtained and conclusions drawn represent a series of "snap-shots" of Site conditions and, while they can be extrapolated to a broad range of conditions, they are most accurate when Site conditions are most similar to those that existed at the time of sampling. In addition, screening criteria do not necessarily reflect the entire range of possible site conditions and, as such, the applicability of conclusions is restricted by these simplifications as well. Identified contributors to these uncertainties are specified below as it pertains to sampling/analysis, exposure, and bioavailability/bioaccumulation/biomagnification/ecotoxicology.

8.2 UNCERTAINTIES IN THE AQUATIC EXPOSURE ASSESSMENT

- Those analytes that were not detected in a medium were excluded from further consideration. Similarly, those COPECs that were detected at less than 5% frequency in exposure media (i.e., surface water and sediments) were also excluded from further consideration. This is a generally accepted practice in ecological risk evaluation in order to prioritize those contaminants that may pose risk. However, it is possible that some compounds were reported as non-detect (ND), but the reporting limit (RL) was above the ecotoxicological benchmarks. These laboratory method limitations introduce some uncertainty in risk evaluation.

- Fish tissue data were not collected for this BERA; instead the data collected for the Krummrich work was used. The fish sampling locations in the Krummrich report do not correspond directly to the sediment and surface water sampling in this report. The Krummrich work was focused around one area, situated between areas R2 and R3 in this report. This source of uncertainty, due to the small area of fish collection, is unlikely to adversely affect the risk evaluation because fish move around the area and fish tissue results are likely to integrate contaminants present throughout the area in various exposure media (surface water, sediments, prey).

- Another source of uncertainty in fish tissue data is that metals were not measured in the Krummrich report, whereas metals were included as target analytes in the analysis of surface water, sediments, and bioaccumulation tests in this study. It is not clear if any Site-related metals are present in fish tissue at high enough concentrations to cause adverse effects.

- The revised work plan called for collecting sediment samples for benthic community identification only if "field observations of collected sediments during the habitat survey indicate that the substrate is substantially different from those sediments observed in the Krummrich work". Based on this, benthic samples were collected from only two locations, one in the Site area (R6AD) and another in the reference (R1BD) area. Thus, in the sediment triad approach, sediment COPEC concentrations and toxicity results could not be compared with the abundance and diversity of benthic community function at all locations.

8.3 UNCERTAINTIES IN THE AQUATIC EFFECTS ASSESSMENT

- There is uncertainty in the selection and use of screening benchmark values for sediment, surface water, and fish tissue data. Surface water benchmarks from Federal (NAWQC), State (IL EPA) or other sources (Tier II SCVs, LCVs) were used. Some of these benchmarks have been developed with limited data, and there are some uncertainties associated with the use of these benchmarks. The guidelines are applied recognizing these limitations, which could tend to either overestimate or underestimate the risks to aquatic organisms.

- The sediment quality guidelines used to screen COPECs must be recognized as very coarse, informal guidelines, and are not necessarily applicable to conditions at the Site. In the case of the sediment quality criteria, none exist against which concentrations of COPECs can be compared to definitively evaluate potential risks. This is because the cumulative uncertainties, inherent measurement errors, and differences between laboratory and *in situ* conditions in current approaches (*e.g.*, apparent effects thresholds, spiked sediment bioassays, equilibrium partitioning) make them far too imprecise for regulatory use as more than very general screening values. In particular, recent efforts by the USEPA to base sediment quality criteria for nonionic organic chemicals on AWQC using equilibrium partitioning have been highly variable, and existing documentation demonstrates that water column species are not necessarily representative of, nor of similar sensitivity to, species inhabiting bottom sediments.

- Several different sediment quality benchmarks were available in the literature and when multiple values were available for a chemical, they were selected and used in a hierarchical approach as described in the work plan. Each of these literature-reported benchmarks were derived for different endpoints and test organisms. In addition, for several compounds, sediment quality guidelines were not available in the literature, and for some of those compounds, benchmarks were derived based on their geochemical properties, toxicity tests and other approaches as described in Section 4.1.2. The lack of availability or the use of derived benchmark values introduces uncertainties in risk evaluation. Therefore, these guidelines, when available, were applied recognizing these limitations, which could tend to overestimate the risks to aquatic organisms.

- Analytical data for river fish are two years old, which may not be indicative of current conditions. Physical/chemical degradation processes, such as biodegradation, volatilization, photooxidation, and/or sorption may have attenuated chemical concentrations, especially organic constituents. Thus, the BERA is conservative.

- Fish body burden-based TRV benchmarks were developed for each contaminant with one specific organism, where different biological effects (e.g., mortality, growth) had been correlated to fish tissue concentrations of the contaminant. Data were not always available for the fish species found at the Site and the measurement endpoints of interest in this study (i.e., no-effect levels on adult fish, growth/reproduction/mortality effects, and whole body analysis). Uncertainties are introduced in the use of other fish species, in the conversion of endpoints (LD_{50} or LOAEL to NOAEL), in the use of surrogate data for some compounds (for 2-methylphenol, alpha-BHC, gamma-Chlordane, endosulfan I, and endrin aldehyde), and in the estimation of TRV from toxicity tests (2,4,5-T; 2,4,5-TP; MCPP). This could tend to overestimate the risks to aquatic organisms.

- Uncertainty in the interpretation of toxicity test results stems from the potential for loss of contaminants and changes in contaminant bioavailability that can occur during sample handling and testing, differences between laboratory and field conditions, and differences between test organisms and the biota that might assimilate the contaminants differently. Although sediments and surface water samples were shipped under low temperature, some of the VOCs could have been lost from test sediments and surface water samples between the time of collection and the time the organisms were exposed to the media. This would possibly underestimate the toxicity of site media to test organisms. Only one VOC compound (1,2-Dichlorobenzene) was detected in sediments at concentrations exceeding sediment quality guidelines (the other analyte, acetone is likely a laboratory contaminant), and no surface water VOCs were above water quality guidelines. Thus, it appears that the potential loss of any VOCs is unlikely to significantly underestimate true toxicity.

8.4 Uncertainties in the Floodplain Exposure Assessment

- Surface soil sampling was generally biased towards areas of known contamination so that estimates of the actual underlying distribution of COPECs made from the data set are conservative. Because the sampling approach was biased towards areas within the five disposal sites where known contamination hot spots where known to occur, there was little opportunity to establish a gradient of ecological risks. As such, the BERA could only characterize risks in areas where contamination was known to occur and could not define areas within the disposal sites where acceptable risks might occur. This has resulted in an overestimation of ecological risks associated with the disposal sites.

- The floodplain exposure assessment assumed 100% bioavailability of COPECs identified in surface soil. This is a significant overestimation of risk as numerous authors have published in the scientific literature regarding the ageing of organic and inorganic constituents and the manner in which these constituents get tied up in soil matrix.

- Only limited surface soil sampling was conducted in each of the disposal areas. Because the sampling was intended to identify and characterize the presence of COPECs in surface soil and not to delineate the boundaries or extent of COPECs, ecological risks for the disposal areas were overestimated.

- Chromium analysis was completed only for total chromium and did not differentiate for hexavalent chromium. Risks may be overestimated as the TRV is based on hexavalent chromium rather than the less toxic trivalent chromium.

- For analytes that were not detected in environmental media, the value of one-half of the detection limit was substituted as a sample concentration. This simplistic technique introduces uncertainty, the direction and magnitude of which cannot be quantified but is conservative because of the probability of including false positives.

- Certain chemicals (*e.g.*, acid extractables) were not detected in most samples but were included as COPECs because of high detection limits. Given the degradation rates of some of these chemicals, it is probable that these constituents are not contained within the environmental media and the biota. Therefore, HQs for these COPECs may overestimate risks.

- Due to limitations in the collection of terrestrial invertebrate, composite samples and assumptions regarding the distribution of the COPECs identified in invertebrate tissue had to be made. This uncertainty could either overestimate or underestimate the ecological risks to shrews.

- ADDs were derived based on the assumption that site-specific food sources used in the model for the ROIs (*e.g.*, shrews and voles for the fox, plants for the prairie vole, fish for mink) comprise 100% of the ROI dietary intake. This does not represent actual field conditions. This assumption may lead to an overestimation of risk as some of the ROIs incorporate other food sources from other areas into their diet.

- The estimation of ADDs in short-tailed shrews was completed using the highest detected concentrations, which assumes that an organism would be exposed to a fixed COPEC concentration at a fixed sample location for its entire life. This conservative approach leads to an overestimation of risk.

- Single values, rather than ranges, have been provided for body weight, ingestion rates, home ranges, and uptake and bioaccumulation factors and incorporated into the ADD calculations. These values represent exposure factors for an average adult receptor and do not account for natural variability or intraspecies differences due to sex, age, pregnancy or lactation. The effect of this uncertainty cannot be quantified nor is the direction of bias able to be determined.

- The dietary ingestion rates for the prairie vole, short-tailed shrew, red fox, mink, and osprey were estimated using allometric regression equations based on body weights (Nagy, 1987; Calder and Braun, 1983) and not from actual experimental measurements or field observations. The effect of this uncertainty cannot be quantified nor is the direction of bias able to be determined.

- ADDs for the red fox were estimated assuming that COPEC body burdens in short-tailed shrews were equal to the prey's COPEC intake (ADD). This uncertainty associated with prey uptake factors have not been established for the red fox; thus, the effect of this uncertainty cannot be quantified nor is the direction of bias able to be determined.

- For those compounds marked as (*), TRVs for structurally similar compounds were used, such as dimethylphthalate for di-n-octylphthalate. The effects of these uncertainties cannot be quantified nor can the direction of the bias be determined.

- Values for the 50% lethal dose (LD_{50}) were used if a LOAEL/NOAEL was not available. The LD_{50} values were treated the same as a subchronic LOAEL with an uncertainty factor of 100 applied. The effects of these uncertainties are conservative and will overestimate risk.

8.5 Uncertainties in the Floodplain Effects Analysis

- Phytotoxicity benchmarks from Efroymson *et al.* (1997) used in the BERA to determine soil HQs are greater than literature trace element concentrations from mature leaf tissue considered normal or sufficient for antimony, cadmium, chromium, mercury, nickel, selenium, silver, and thallium (Allen *et al.*, 1995; Pendias and Pendias, 1992). Thus, the risk estimates for these metals may be overestimated.

- COPEC screening phytotoxicity benchmarks derived by Efroymson *et al.*, (1997) are, by admission of the authors, highly conservative and not necessarily predictive of site-specific effects.

- Given that data were either rejected or no analysis was completed for certain compounds for some areas, the ADDs for some COPECs may not be complete. Data were rejected for the following:

Acetone was rejected for soil sample Soil-O-1; 4,4,DDD and alpha-Chlordane was rejected for Soil-Q-2, and Soil-Q-4; and 4,4 DDD was rejected for sample number Soil-Q-12. 4,4 DDT, and all the herbicides except for Dinoseb were rejected for pond sediment sample number P11S. 2,4, Dimethylphenol was rejected for the following insect tissue samples and plant samples: IN Q1, IN Q2, IN OS1, IN P1, IN Q1, IN Q2, IN ROS1, PL-OS-2, PL-OS-3, PL-OS-4, PL-O-1, PL-O-2, PL-O-3, PL-P-1, PL-P-2, PL-P-3, PL-P-4, PL-R-1, PL-R-2, PL-R-3, PL-R-4, PL-S-1. All herbicides except for 2,4,5-TP and Dinoseb were rejected for the worm sample #13 (Site Q-16).

- TRVs were estimated using NOAELs/LOAELs primarily derived from laboratory animals. Laboratory animals are bred for their chemical sensitivity and the environment in which laboratory animals are kept, as well as the manner in which they may be dosed (*e.g.*, gavage) does not reflect *in situ* conditions. Thus, TRVs most likely overestimate risks.

  NOAEL/LOAELs were not available for most wildlife receptors but were estimated by:

  - Extrapolation from laboratory rodents (mice, rats) to wild rodents (shrews, prairie voles) and other mammals (fox)
  - Extrapolation of NOAELs/LOAELs from different feeding niches [*i.e.*, granivorous birds (domestic chickens, Japanese quail) or omnivorous birds (blackbird) to the piscivorous osprey];
  - Extrapolation from NOAELs to LOAELs and vice versa; and
  - Extrapolation from acute/LD_{50} or subchronic studies to chronic studies.

- Wildlife TRVs are conservative because the lowest NOAEL/LOAEL benchmarks in the scientific literature were chosen if multiple doses were provided.

- HQs from silver in terrestrial receptors may be overestimated due to low half-life persistence and rapid clearance from the body.

- NOAEL/LOAEL benchmarks are single points that infer that all organisms will have adverse effects above that concentration. It does not account for individual or population variability. The effects of this uncertainty cannot be quantified nor the direction of bias determined.

8.6 UNCERTAINTIES WITH ECOTOXICOLOGY

- The analysis performed for this assessment did not account for site-specific factors such as adaptive tolerance, reproductive potential, the small size of the affected areas, and recruitment from similar adjoining areas. Such factors would tend to mitigate the degree and ecological significance of loss or impairment of a portion of ecological population(s) due to both chemical and physical stressors in the area. As a result, the approach used in this assessment necessarily results in overestimation of risk.

- ADDs were calculated based on 100% bioavailability and do not account for various degradation processes that may affect concentrations within the environmental media. The chemical concentration available for systemic circulation to the target organ is considerably smaller than the environmental concentration; thus, risk estimates are overestimated.

- HQs and HIs based on low molecular weight SVOCs (*i.e.*, pentachlorophenol) and some metals (e.g., barium, chromium, cobalt, copper) are overestimated because scientific literature indicates that these compounds do not bioaccumulate/biomagnify but quickly volatilize or biodegrade or are rapidly metabolized and excreted from the organism.

- Uncertainties regarding the assessment endpoints (*i.e.*, regulatory criteria such as the AWQC sediment quality guidelines) derive principally from the fact that they are conservative and generic and protective of all species, including the most sensitive, rather than site-specific indicators of potential risk to ecological receptors present at the Site. AWQC are derived from the highest quality and most applicable data that were available at the time of development. However, the physiology and toxicology/pharmacology of COPECs, particularly in relation to the wildlife taxa identified as ROIs, is known only with some certainty. This conservatively biased uncertainty will affect risk estimates in a manner that cannot be quantified, but incorporates both error and biological variation. The latter is likely the largest component of uncertainty, because organisms have highly variable responses to toxicants, and extrapolation from laboratory studies to field exposure estimates incorporates this conservatively biased uncertainty into the risk estimates.

Section 9

9.0 RESULTS AND CONCLUSIONS

This BERA evaluated the potential for adverse ecological impacts to occur as a result of the exposure of receptors found within the Mississippi River adjacent to the Area 2 Sites and the floodplain area containing the Area 2 Sites to COPECs originating from five disposal sites. To assess the potential for adverse ecological impacts to occur, six assessment endpoints were evaluated. The following conclusions can be drawn based on the completion of this BERA.

9.1 POTENTIAL AQUATIC ECOLOGICAL RISKS IN THE MISSISSIPPI RIVER

Potential ecological risks to aquatic receptors within the Mississippi River were assessed through the collection of surface water and sediment samples from locations upstream, adjacent to, and downstream of the five disposal Sites. The samples were chemically analyzed to determine the concentrations of COPECs possibly present. Bioassays were run on both surface water and sediment samples to evaluate acute and chronic toxic effects to the endpoint species. Additionally, bioaccumulation tests were conducted to determine the body burdens of COPECs in test organisms exposed to sediments for an extended period of time. Fish tissue body burdens identified in historic sampling activities were also evaluated to assess potential ecological impacts.

An Interim Groundwater Remedy is currently being implemented downgradient of Sauget Area 2 Sites O, R and S to control adverse impacts on the Mississippi River due to groundwater discharges from these Sites; Sauget Area 1 Sites G, H, I and L; and industrial facilities in Sauget and Cahokia, Illinois.

The results of the BERA for the different evaluated media are presented below. Table 9-1 summarizes the COPECs identified in various aquatic media by this BERA.

Sediments – The BERA concluded that there were no adverse ecological impacts associated with the presence of COPECs in sediments.

Chemical analysis of sediments indicated that there were measurable concentrations of COPECs that exceeded conservative ecologically based benchmarks. The COPECs included volatile organic compounds (VOCs) such as acetone and chlorobenzene; semivolatile organic compounds (SVOCs) such as 1,2-dichlorobenzene; pesticides such as dieldrin, endrin aldehyde, heptachlor epoxide; herbicides such as MCPP; and metals such as arsenic, barium, cadmium, copper, lead, manganese, nickel, and zinc. The highest detected concentrations of organic COPECs were located along transects closest to the shore in the sampling area located downgradient of Site Q (North) and just downstream of Site R. None of the inorganic COPECs exceeded their respective benchmarks by a significant degree and the pattern of distribution throughout the sampling plots adjacent to or downstream of the Sauget Area 2 Sites appeared to be random.

However, the sediment bioassays (considered to be a stronger indicator of potential toxic effects) demonstrated that there were no significant toxic effects in any of the Site-related sediment samples. For the acute toxicity test, there were no significant differences in mean survival when Site-related samples were compared to their respective control samples for any of the sampling sites adjacent to, or downstream of, the disposal areas. Similarly, the chronic test concluded that none of the sediment samples collected from any of the sampling plots exhibited mean growth that was significantly lower than the mean growth of the corresponding laboratory control samples.

Surface Water - The BERA concluded that there were limited ecological impacts associated with the presence of COPECs in surface water.

Surface water COPECs identified through chemical analyses included p-chloroaniline, 2,4-D, aluminum (total), barium (dissolved, total), copper (total), iron (total), manganese (total), and vanadium (dissolved, total). P-chloroaniline had the greatest exceedance of its conservative screening benchmark, followed closely by 2,4-D. Maximum concentrations of these two constituents were detected at the sampling area downgradient of Site Q (North) and just downstream of Site R on the transects closest to the riverbank. Barium had the greatest exceedance of its benchmark, while the remaining metals only slightly exceeded their respective benchmarks.

Surface water bioassays indicated that acute toxicity was limited to the sampling area downgradient of Site Q (North) and just downstream of Site R. The sample with the lowest survival and young production corresponded to the surface water sample that had the highest concentrations (by nearly an order of magnitude) of p-chloroaniline and 2,4-D. Chronic toxicity was also seen at other sampling locations downstream where detected concentrations of p-chloroaniline and 2,4-D were noted.

Aquatic Risk Assessment Conclusions - The BERA concluded that no adverse ecological impacts were identified with sediments within the Mississippi River and only limited surface water impacts were identified. Two organic compounds (p-chloroaniline and 2,4-D) were identified as the principal constituents of concern in the surface water environment of the Mississippi River adjacent to the Sauget Area 2 Sites.

Historical sampling performed at Sauget Area 2 Site R, which is immediately upstream of Sampling Area R3, indicates that p-chloroaniline and 2,4-D are present at this site. Sediment and surface water sampling performed by Menzie-Cura in October and November 2000 indicated that groundwater discharging to surface water downgradient of Site R resulted in an adverse impact on the Mississippi River. Based on this information, USEPA issued a Unilateral Administrative Order (Docket No. V-W-'02-C-716) on September 30, 2002 for performance of an Interim Groundwater Remedy, consisting of installation of a physical barrier and groundwater extraction system downgradient of Site R, to protect the Mississippi River. Groundwater extraction started on July 15, 2003, and construction of the physical barrier is scheduled to start on September 2, 2003 and be completed in the first quarter of 2004. The implementation of the interim groundwater remedy will eliminate the discharge of contaminated groundwater into the river. This will eliminate the potential ecological risks identified with these two compounds. For that reason, no additional remedial action is considered necessary to protect the aquatic ecosystem in the Mississippi River.

9.2 POTENTIAL ECOLOGICAL RISKS IN THE FLOODPLAIN

The BERA evaluated the potential for COPECs to impact Receptors of Interest (ROIs) with small home ranges (prairie vole and short-tailed shrew) and large home ranges (osprey, mink and red fox). Potential for adverse impacts was evaluated on a site-by-site basis for the vole and shrew because of their small foraging areas and on study area basis for the osprey, mink and fox because of their large foraging areas. For the small-ranging organisms at the individual sites, the prairie vole was considered the most appropriate indicator of potential ecological risks because habitat suitable to support the short-tailed shrew was not dominant at the five disposal areas (Sites O, P, Q, R and S). Risks to these organisms were calculated based on food chain models using concentrations of COPECs identified in surface soil, plant tissues, and invertebrate body burdens as input parameters.

Potential floodplain ecological risks are summarized below.

Piscivores - A limited number of COPECs were identified for consumption of fish and surface water by the mink and osprey, two organisms that were evaluated based on aquatic exposures. From a habitat standpoint, the riverbank adjacent to the Sauget Area 2 Sites is not good habitat for any fish-eating mammal. Much of the 14,000 linear feet of riverbank is covered with stone riprap, removing cover requirements that this animal has. The remainder of the bank contains piers, pilings, buildings and other human disturbances, which would further preclude fish-eating mammals from inhabiting the area.

Nitrobenzene, MCPP, PCBs, dioxin/furans, aluminum and antimony were all identified as COPECs for the mink. However, most of the estimated ecological risks for the mink were based on consumption of fish from the large pond. The large pond is one of two ponds located in the southern end of Site Q. Identified as Site Q (Ponds), these ponds are ephemeral water bodies that will support a fish community on a temporary basis only if fish are washed into the ponds through overbank flooding of the Mississippi River. Fish were collected from the large pond, prior to it's drying up, and analyzed for the presence of COPECs. If those fish are removed from the modeling, as the community no longer exists, then the only COPECs identified for the mink are MCPP and antimony. The adverse risks noted with those constituents were slight.

For the osprey, mercury was the only COPEC. The potential for an ecological risk was small. Since surface water concentrations and bioaccumulation factors were used to calculate fish tissue mercury concentrations, actual risks due to mercury are likely to be lower than the predicted risks.

Plants - The potential for direct impact to plants was evaluated by comparing surface soil concentrations to screening plant benchmarks. A variety of COPECs in each disposal site were identified with concentrations in excess of these benchmarks. Site S had the highest number of organic COPECs that exceeded the plant benchmarks, while Site Q had the highest number of inorganic COPECs in excess of the conservative screening plant benchmarks.

These benchmarks are considered to be highly conservative even by the authors of the benchmarks. While a number of COPECs were identified, no indication of impacts to plants was noted in field observations conducted at the Sites. The vegetative communities in each of the disposal areas were marked by robust and vigorous plant growth with no indications of phytotoxic effects. The prairie vole food chain model provides a more accurate assessment of potential plant impacts by evaluating the presence of COPECs that were identified in plant tissues as they relate to a higher trophic level receptor.

Herbivores - In examining the potential for ecological risks at the five disposal sites, no risks were identified at Site P or Site Q (South) for the prairie vole. Potential ecological risks were predicted at Site O (PCBs, dioxin/furans, mercury, and thallium) and Site S (pentachlorophenol, PCBs, and mercury). At Site O, only PCBs and dioxins/furans exceeded both the NOAEL and the LOAEL benchmark values for the prairie vole. Potential areas of ecological risk at Site O are centered on sampling locations W-O-1 and W-O-3 and are shown on Figures 9-1 through 9-3. Adverse risks were also predicted for Site R (cobalt and mercury), however Site R is covered with a dirt cap. Further, the cap is regularly mowed and, consequently, is not considered a viable habitat for the vole. The potential adverse risks estimated at Site R were not considered to be significant.

Carnivores - An assessment of the potential for site-wide adverse ecological impacts to the red fox were conducted to determine whether cumulative affects from the five disposal Sites would be noted. The assessment was made based on modeled exposure to prey items (the short-tailed shrew and the prairie vole). In keeping with OSWER Directive 9285.7-28P, as an upper trophic level organism, the red fox was considered be the more critical receptor while the importance of the two small mammals was as prey items.

Aluminum had the highest exceedance of both its NOAEL and LOAEL benchmark values. PCBs and dioxin/furans, which were expected to be in prey tissue based on the model parameters, also exceeded their NOAEL and LOAEL benchmarks. Site O and Site S were the only sites where PCBs were modeled to be present at elevated concentrations (in excess of TRV benchmarks) in both the shrew and the vole and these two sites served as the greatest contributor of PCB and dioxin/furan risks to the red fox. Since the risks for PCBs were predicted based on the shrew and the vole as a prey base for the fox, areas potentially needing remedial action to protect these organisms from PCBs and dioxins/furans would also potentially protect the red fox. These areas are shown on Figures 9-2 and 9-3.

It is noted that the red fox has a mean home range of 1,727 acres; it is highly unlikely that the disposal sites (total area less than 150 acres) would support a large population of red fox. Noting the discontinuity of the sites, it is more likely that a small number of fox utilize a portion of different disposal areas for foraging, moving between contaminated and non-contaminated areas. Additionally, the fence surrounding Site R would limit access of the fox to this disposal area.

Ponds - The BERA also evaluated potential ecological risks to aquatic receptors associated with the aforementioned ponds. While sediment and surface water screening against conservative benchmarks indicated the presence of some organic and inorganic COPECs, acute and chronic toxicity testing of both matrices did not indicate any adverse effects. However, as the ponds were mostly dried by the time this BERA was implemented and only a partial data set could be collected to evaluate them. Surface water and sediment quality data were collected in June 2003. These data will be presented in an addendum to this BERA at a future date.

Floodplain Risk Assessment Conclusions - The BERA identified the potential for adverse ecological impacts associated with the presence of COPECs in surface soil found in Site O and Site S. For Site O, the most significant COPECs included dieldrin, lindane, PCBs, dioxins/furans, aluminum, and mercury. For Site S, the most significant COPECs included pentachlorophenol, beta-BHC, endrin, and lindane, and PCBs. These areas will be evaluated further in the Feasibility Study for the identification of potential remedial actions. Limited ecological risks were identified with surface water and sediments in Site Q (Ponds), however, a further determination of potential ecological risk will be made upon the evaluation of surface water and sediment quality data collected in June 2003.

Section 10

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Appendix I-A

Ecological Risk Assessment Work Plan

12.0 ECOLOGICAL RISK ASSESSMENT WORK PLAN

This section outlines the approaches and methodologies to be used in the preparation of the ecological risk assessment (ERA) for the Sauget Area 2 Sites (the Sites) located in the Villages of Sauget and Cahokia, Illinois. Environmental concerns at the Sites are being addressed subject to an Administrative Order by Consent between the Sauget Area 2 Sites Group (the Group) and the U. S. Environmental Protection Agency (USEPA) Region V, pursuant to Section 106 of the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). The Sites include five identified source areas (Sites O, P, Q, R, and S). Additionally, the Sites front approximately 8,000 linear feet of the Mississippi River.

This work plan was developed based on the following guidance material:

- *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA 540-R-97-006, June 1997);
- *Guidelines for Ecological Risk Assessment* (EPA/630/R-95/002F, April 1998);
- *Framework for Ecological Risk Assessment* (EPA630/R-92/001);
- *Developing a Work Scope for Ecological Assessments* (Eco Update, Volume 1, Number 4, May 1992);
- U.S. EPA Region V Ecological Assessment Guidance;
- *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 540 G-80 004, October 1988); and
- *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (OSWER Directive 9285.7-28 P).

12.1 Scope and Objectives

The objective of the ERA is to evaluate the potential for adverse ecological effects to occur as a result of exposure to Site-related constituents by biological receptors living within the aquatic and terrestrial ecosystems located on or adjacent to the Sites. The ERA will be a baseline evaluation of ecological risks utilizing both historic data regarding the Sites and data to be collected as part of future planned investigative activities within the Mississippi River and the five Sites. The ERA will be prepared using conservative, but realistic, assumptions about potential exposures and, since it is a baseline assessment, will assume no remedial action has occurred.

Specifically, the principal functions of the ERA described in this work plan are:

- Determine whether actual or potential ecological risks currently exist at the Sites;
- Identify those constituents present at the Sites that pose potential ecological risks; and
- Generate data and information for risk management and risk reduction decisions.

This work plan outlines the procedures by which data collected from aquatic and terrestrial sections of the Sites will be evaluated to assess the potential for adverse ecological effects. This ERA will evaluate current site data that will be collected through the following planned activities:

1. *Aquatic Systems*

- Chemical analyses of sediment samples collected from the Mississippi River and on-Site ponds;
- Community evaluation (species richness and total biomass) of benthic macroinvertebrates collected from the Mississippi River and on-Site ponds (should they be present);
- Bioassay analyses of toxicity tests conducted on sediment collected from the Mississippi River and on-Site ponds;
- Bioaccumulation studies on sediment samples collected from the Mississippi River and on-Site ponds;
- Chemical analyses of surface water samples collected from the Mississippi River and on-Site ponds;
- Bioassay analyses of toxicity tests conducted on surface water collected from the Mississippi River and on-Site ponds;
- Chemical analyses of fish tissue collected from the Mississippi River and on-Site ponds; and

- Observations of the fish community and feeding habits of fish collected from the Mississippi River and on-Site ponds.

2. *Terrestrial Systems*

- Chemical analyses of collocated surface soil, plant tissue, and earthworm samples collected from the five Sites.

The specific details of how these data will be collected are presented in separate Field Sampling Plans (FSPs) and Quality Assurance Project Plans (QAPPs). Specific documents have been developed for the Mississippi River (Volume 3A and 3B of the Support Sampling Plan) and terrestrial portions of the Sites adjacent to the river (Volume 4A and 4B of the Support Sampling Plan). It is the objective of this Ecological Risk Assessment work plan to describe how the data collected as per the FSPs and QAPPs will be evaluated to assess the potential for ecological risks associated with constituents of concern in the Mississippi River and the terrestrial portions of the Sites.

12.2 Work Plan Organization

This work plan is divided into the following sections:

- Ecological Risk Assessment Process;
- Ecological Setting;
- Selection of Chemicals of Potential Ecological Concern (COPECs);
- Identification of Receptors and Endpoints;
- Ecotoxicological Benchmarks;
- Wildlife Exposure Models;
- Risk Characterization;
- Uncertainties; and
- Report Preparation

12.3 Ecological Risk Assessment Process

USEPA has developed and issued structured guidance for conducting ERAs. In 1992, USEPA presented a general framework for conducting ERAs that outlined the concepts of assessment and measurement endpoints (USEPA, 1992a). The framework document was intended to be the first step in the promulgation of a simple and flexible structure for evaluating the potential for ecological risks within the USEPA. The framework document outlined the completion of an ERA in terms of:

- Problem Formulation – This is the first phase of the ERA during which the goals, breadth, and focus of the assessment are articulated;

- Analysis – The analysis phase consists of the technical evaluation of data. This phase is divided into the characterization of exposure and the characterization of ecological effects; and

- Risk Characterization – During this phase, the likelihood of the expression of adverse effects resulting from the exposure of a receptor to a stressor is evaluated.

This framework approach was further defined in 1998 with the publishing of USEPA's general guidelines for conducting ERAs (USEPA, 1998a). USEPA (1998a) placed new emphasis on ensuring that the results of the assessment can be used to support risk management decisions.

Almost concurrent with the issuance of the 1998 guidance document, an interim final programmatic guide to the development of ERAs under CERCLA was developed by the USEPA Office of Emergency & Remedial Response (USEPA, 1997a). This guide (Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments or "ERAGs") placed the three phases of the ERA process into a more structured eight-step process for the development of ERAs specifically at CERCLA sites. This allowed for a more proactive mechanism to measure the progress and organization of the ERA. The eight steps outlined in that document (and applied to the ERA being prepared for the Sites) are:

- Step 1 – Preliminary Screening Level, which includes, preliminary problem formulation, and preliminary toxicity evaluation.

- Step 2 – Screening Level, which includes development of exposure estimates and preliminary risk calculations. The step includes a Scientific/Management Decision Point **(SMDPa)**.

- Step 3 – Baseline Risk Assessment Problem Formulation, which includes toxicity evaluation, development of a preliminary site conceptual model and exposure pathways, and development of assessment endpoints. This step also includes a **SMDP (SMDPb)**.

- Step 4 – Study Design and DQO Development. This step includes development of the Work Plan, and Sampling and Analysis Plan based upon results of the previous three steps. This step also includes an **SMDP (SMDPc)**.

- Step 5 – Verification of Field Sampling Design. a site visit This step includes a determination of the feasibility of the field program as outlined in Step 4. This step includes an **SMDP (SMDPd)**.

- Step 6 - Site Investigation and Data Analysis. This step includes an **SMDP**.

- Step 7 – Risk Characterization. This step includes more refined and detailed quantification of potential site risks, and is generally a more realistic evaluation of risks than was performed in Step 2.

- Step 8 – Risk Management, which includes selection of alternatives in the Record of Decisions as a **SMDP (SMDPe)**.

SMDPs are checkpoints in the ERA process to:

- Verify that the work that was conducted at each step is complete;

- Determine whether the risk assessment is proceeding in a direction that will support decision making; and

- Determine the need, if any, for proceeding to the next step.

SMDPs provide the opportunity to further focus assessment approaches or add additional activities to address the specific goals of the ERA. They also provide the opportunity to exit the process where the weight-of-evidence supports no further action, since all eight steps may not be required for all site evaluations.

USEPA has also issued a set of risk management principles that are relevant to ERAs and that serve to supplement the ERA guidance (USEPA, 1999a). This directive, prepared by the Office of Solid Waste and Emergency Response (OSWER), recommends the following series of risk assessment/risk management questions be answered at each SMDP:

- <u>What ecological receptors should be protected?</u> Site-specific assessment endpoints should be identified that address chemical-specific potential adverse effects to local populations and communities of plants and animals. The role of structure and function of the endpoint becomes important in this decision (Keenan et al., 1999). For example, the structure of the benthic community itself (i.e., its diversity) may be less important to the local system than the higher trophic level it supports (i.e., its function as a food source for higher trophic level organisms).

- <u>Is there an unacceptable ecological risk at the site?</u> Ecological impacts can be readily apparent (e.g., loss of vegetation) or less apparent (e.g., slight change in benthic abundance). A variety of assessment and measurement endpoints may be needed to generate lines-of-evidence to determine whether a potential exists for an unacceptable ecological risk. It is also important to determine whether or not the observed "effect" is due to site-related constituents or from indigenous conditions (e.g., naturally reducing conditions causing a paucity of benthic organisms).

Remaining ecological risk assessment/risk management questions discussed in USEPA (1999a) emphasize issues related to remediation. However, they need to be kept in mind as the investigation and ecological risk assessment for the Sites are completed.

- Will the cleanup cause more ecological harm than current site contamination? Short-term and long-term effects of the contamination should be considered, as well as the ability of the system to recover from any disturbance related to remediation. For example, it may be counterproductive to remove a bottomland hardwood forest based on a presumed short-term ecological impact to a small mammal when the forest cannot be restored due to issues related to water management as well as the long period of time required for system recovery.

- What cleanup levels are protective? If a decision is made that a remedial action is required, the various lines-of-evidence that are used to evaluate the potential risks are revisited to determine an appropriate cleanup goal. The likelihood of the response alternatives to achieve success and the time frame for an ecological community to fully recover should be considered in the remedy selection. Generally, empirical data supporting a concentration/response gradient is the most appropriate to use for this assessment.

In addition to the above decisions, the OSWER Directive identifies six principles that risk managers should address when scoping ecological risk assessments or when making ecological risk management decisions (USEPA, 1999a). The principles are:

- Principle Number 1 – Reduce ecological risks to levels that will result in the recovery and maintenance of healthy local populations and communities of biota.

- Principle Number 2 – Coordinate with Federal, State, and/or Tribal Natural Resource Trustees.

- Principle Number 3 – Use site-specific ecological risk data to support cleanup decisions. Site-specific data are used to determine whether or not site releases present unacceptable risks and to develop quantitative cleanup levels that are protective.

- Principle Number 4 – Characterize site risks in terms of magnitude (i.e., the degree of observed or predicted responses of receptors to the range of contaminant levels), severity (i.e., how many and to what extent the receptors may be affected), distribution (i.e., aerial extent and duration over which effects may occur), and the potential for recovery of the affected receptors.

- Principle Number 5 – Communicate risks to the public.

- Principle Number 6 – Remediate unacceptable ecological risks.

One of the critical points in this memorandum is that USEPA has directed the ecological risk assessment process to examine populations, as opposed to individuals.

12.3.1 Consistency with the ERAGs Process

ERAGs (USEPA, 1997a) were used as a basis for the development of this ERA Work Plan. However, because of USEPA's desire to expedite certain aspects of the Remedial Investigation/Feasibility Study (RI/FS) process for the Sites, adjustments were made in the ERAGs process. A limited Step 1 (Screening Level Problem Formulation and Ecological Effects Evaluation) and Step 2 (Screening Level Exposure Estimate and Risk Calculation) have been conducted for a section of the Mississippi River as part of other environmental studies. A screening level assessment has been conducted for the terrestrial portion of the Sites on a qualitative basis using some historic data and assessment reports prepared by various regulatory agencies.

This Work Plan outlines the procedures to be used in the development of a Baseline Ecological Risk Assessment (BERA) for the aquatic and terrestrial communities at the Sites. This Work Plan continues the ERAGs process by detailing Step 3 (Baseline Risk Assessment Problem Formulation) in Sections 12.4 and 12.5 (identification of exposure pathways and conceptual site model), 12.6 (identification of chemicals of concern), and 12.7 (identification of assessment endpoints). Step 4 (Study Design and DQP Process) is outlined in Section 12.7 (identification of measurement endpoints and lines of evidence), Section 8 of this SSP and Volumes 3A, 3B, 4A, and 4B (work plan and sampling and analysis plan). Step 5 (Verification of Field Sampling Design) will be made during the preliminary site reconnaissance described in Volumes 3A, 3B, 4A, and 4B. Step 6 (Site Investigation and Data Analysis) will be completed during the implementation of the various sampling plans and the completion of Step 7 (Risk Characterization) is described in Section 12.8.

Regarding the use of SMDPs, discussions held with the USEPA, U.S. Fish and Wildlife Service, Illinois Environmental Protection Agency, and the government contractor served to consolidate SMPDa, SMDPb, and SMDPc. Discussions to be held following the preliminary site reconnaissance will serve as SMDPd.

12.3.2 Consistency with the DQO Process

The DQO process is a seven step, iterative planning approach used to prepare plans for the collection of environmental data. It will provide a systematic approach for determining the criteria that a sampling program should fulfill, the procedures to be used in the collection of the samples or measurements, determine tolerable error rates, and identify the number of samples or measurements that should be made (USEPA, 2000).

In keeping with the ERAGs process (USEPA, 1997a), Step 3 (Baseline Risk Assessment Problem Formulation) and Step 4 (Study Design and DQP Process) comprise the DQO process for the development of ERAs. As previously mentioned, Step 3 is described in Sections 12.4 and 12.5 (identification of exposure pathways and conceptual site model), 12.6 (identification of chemicals of concern), and 12.7 (identification of assessment endpoints). Step 4 is outlined in Section 12.7 (identification of measurement endpoints and lines of evidence), Section 8 of this SSP and Volumes 3A, 3B, 4A, and 4B (work plan and sampling and analysis plan). Further detail on the DQO process used in the development of sampling/analytical strategies is presented in Volumes 3A, 3B, 4A, and 4B.

12.4 Ecological Setting

The Sites are situated adjacent to the Mississippi River. The Sites, found within the villages of Sauget and Cahokia, Illinois, are generally located east of the Mississippi River, south of the MacArthur bridge railroad tracks, west of Illinois State Highway 3, and north of Cargill Road. The Sites front approximately 8,000 feet of the Mississippi River.

The Sauget Area 2 Sites includes five disposal areas, Sites O, P, Q, R, and S, adjacent, or in close proximity, to the Mississippi River. These five disposal areas were given letter designations by the Illinois Environmental Protection Agency (IEPA) in the 1980s. Two of these sites, Sites Q and R, are located on the wet side of the flood wall and levee that is operated and maintained by the US Army Corps of Engineers and the Metro East Sanitary District. The flood wall is designed to protect the City of East St. Louis and the Villages of Sauget and Cahokia from flooding. Sites O, P, and S are located on the dry side of the flood wall and levee.

Site O is located on Mobile Avenue in Sauget and occupies approximately 20 acres northeast of the American Bottoms Regional Wastewater Treatment Facility (ABRTF). Site P is located between the Illinois Central Gulf Railroad and the Terminal Railroad, and is north of Monsanto Avenue in the Village of Sauget. Site P occupies approximately 20 acres of land. Site Q occupies approximately 90 acres and is south of Sauget Site R and the old Union Electric Power Plant, west of the Illinois Central Gulf Railroad and the U.S. Corps of Engineers flood control levee, and east of the Mississippi River. The two ponds created by the borrow pit operations are located at the southern end of Site Q. Site R is located adjacent to the Mississippi and has had a temporary cap placed on it. Site S is a small disposal area west-southwest of Site O.

The following sections provide the basis for the problem formulation stage of the ERA in that the potential pathways and receptors are identified for future evaluation. The ecological condition section provides a general understanding of the ecological receptors and communities found within the Sites. At present, this is a general presentation of information as detailed evaluations have only been completed on a narrow corridor adjacent to the Sites (i.e., the Dead Creek corridor). Information detailing the characteristics of the ecosystems in each of the five Sites and in the Mississippi River adjacent to the Sites will be identified and compiled as part of the field data collection activities.

During the course of the field activities supporting the ERA, a more thorough understanding of the characteristics of the aquatic and terrestrial ecosystems associated with the Sites will be developed. Aquatic sampling will provide details of the fish and benthic invertebrate communities, as well as physical characteristics of the surface water and sediments in the Mississippi River adjacent to the Sites. During the terrestrial sampling, a description of the habitat and dominant vegetative communities will be developed. This plant community survey will include a determination of community makeup, density, frequency and abundance. The objective of that activity will be to build a general habitat cover type map for the Sites within the Sites and to prepare a basic inventory of the dominant plant and animal species indigenous to the area.

12.4.1 Ecological Conditions

Sauget Area 2 is located in the floodplain of the Mississippi River in an area known as American Bottoms. Topographically, the area consists primarily of flat bottomland, although local topographic irregularities do occur. Generally, land surface in the American Bottoms slopes from north to south and from east to west, toward the Mississippi River. Land surface elevation ranges from 400 to 410 feet above Mean Sea Level (MSL) with little topographic relief.

The dominant ecological feature associated with the Sites is the Mississippi River. The floodplain area containing the Sites fronts approximately 14,000 linear feet of the Mississippi River. The terrestrial area is found within a category of ecosystems called floodplains, which are flat land surfaces where alluvial material has been laid down in old valleys over an extended time period. While anthropogenic influences have greatly modified the Mississippi River and the quality of its basic components, the river still influences the types of sediments and soils found near the Site, the types of aquatic organisms found within its influences, and the basic structure for the terrestrial habitat found adjacent to its shores.

The section of the Mississippi River adjacent to the Sites is called the Unimpounded Reach (USGS, 1999), which extends from St. Louis downstream to Cairo, Illinois. This section of the river, also called the Open River Reach, is characterized by channelized aquatic habitats, with terrestrial portions that have been protected from flooding by levees to support agriculture and other uses of the historic floodplain. This scenario reflects the conditions adjacent to the Site as the riverbank has been substantially sculpted by anthropogenic activities. A rock revetment covers approximately 2,250 feet of the riverbank adjacent to Site R, and the remainder of the riverbank has been developed as piers and other structures for barge traffic. In this reach of the Mississippi River there are almost no lotic or marsh habitats. Channel depth in the center of the channel is maintained at a minimum depth of nine feet to allow for barge traffic. The current is swift, with median flows ranging from 85,000 to 95,000 ft^3/second.

USGS (1999) notes that the characteristics of sediments and surface water in the Mississippi River below its confluence with the Missouri River (approximately two miles upstream of the Sites) have long differed from the reach upstream of the confluence. Two of the prominent reasons for those differences include both the City of St. Louis and the drainage basin of the Missouri River. St. Louis has had a significant effect on water quality within the river due to sewage and industrial discharges from within the city. Missouri Department of Natural Resources (1994) notes that an estimated 300 tons of ground garbage was discharged in the river daily in 1957, and as late as 1970, raw sewage was discharged directly into the river by the City of St. Louis (Corbett, 1997). The Missouri River drains an area with highly erodible soils and is the major source of sediments to the Mississippi River. This contribution of sediment leads to changes in water clarity, sedimentation of shallow areas, and the introduction of non-Site-related sediment-borne constituents.

Boyer (1984) notes that St. Louis contributes significant amounts of constituents from wastewater effluents, industrial discharges and urban runoff, including metals and organic compounds, such as PCBs. Pesticides and herbicides are significant contaminants in the Mississippi River. The reach upstream of the Missouri confluence contributes 40 to 50 percent of the pesticide and herbicide load within the Mississippi River, even though it represents only 22 percent of the flow from the entire river (Goolsby and Pereira, 1995). USGS (1999) notes that surface water conditions in the Mississippi River have improved since the passage of the Federal Water Pollution Control Act in 1970, though concentrations of pesticides still exceed USEPA guidelines during low flow, high use periods of the year. Such concentrations of metals and organic compounds (that often exceed screening benchmarks) have the potential to affect biological communities.

Aquatic life within the river depends upon the presence of suitable habitat, which is a function of water and sediment characteristics. Areas of deep, swift water, such as found adjacent to the Sites, would be occupied by channel dwelling fishes and would probably not support habitat that would be used for spawning or as nurseries. Fremling et al. (1989) state that the Upper Mississippi River Basin supports at least 260 freshwater fish species. Fish in channel habitats are called riverine species and occur as either streamline forms that occupy the water column such as white bass (*Morone chrysops*) or bottom-dwelling forms, such as channel catfish (*Ictalurus punctatus*) (USGS, 1999). Other common riverine species identified by USGS (1999), based on Long-Term Resource Monitoring Program (LTRMP) catch data include sauger (*Stizostedion canadense*), walleye (*Stizostedion vitreum*), and smallmouth buffalo (*Ictiobus bubalus*). Important prey species indigenous to the Unimpounded Reach area include gizzard shad (*Dorosoma cepedianum*) and emerald shiners (*Notropis atherinoides*).

While a survey of the terrestrial community in each of the Sites will be conducted, preliminary observations indicate that Sites O, P, R and S have been significantly impacted by anthropogenic activities. These activities include clearing and construction of roads and railroad lines, construction of buildings, and the development of industrial activities. These areas show signs of extensive clearing and/or disturbance, and they are vegetated either solely by herbaceous communities, or by herbaceous communities with a thin layer of early successional shrubs or trees.

Site Q is the one area having a significant quantity of floodplain forest still in evidence. USGS (1999) notes that floodplain forests are more structurally complex than upland forests and are generally differentiated into three strata. Those strata include an herbaceous ground cover layer, a shrub or sampling sub-canopy layer, and a tree layer that dominates the community. The major floodplain forest communities in the Upper Mississippi River System include those dominated by black willow (*Salix nigra*), those dominated by eastern cottonwood (*Populus deltoides*), those dominated by silver maples (*Acer saccharinum*) and those dominated by a mixed oak-hickory forest.

In 1997, a biological survey was conducted at select locations of Site Q as part of an Ecological Risk Assessment conducted by Ecology and Environment on behalf of the USEPA (USEPA, 1997b). The focus of the ERA was a small pond located in the northern portion of Site Q, and the two ponds located at the southern end of this Site. The ponds at the southern end have been identified as a location for sampling as part of the field activities to support this ERA. The northern pond was noted as being devoid of life, though the southern ponds did support populations of aquatic macrophytes and an amphibian (chorus frog, *Pseudacris triseriata*). Subsequently, a brief reconnaissance of the area in January 2001 noted that the two southern ponds were dry and did not contain any standing water. However, anecdotal statements made by USEPA personnel in April 2001 indicate that the ponds have refilled as a result of stormwater influence.

Plant species identified by the USEPA in their survey were cocklebur (*Xanthium strumarium*), common mullein (*Verbascum thapsus*), common evening primrose (*Oenothera biennis*), black-eyed susan (*Rudbeckia serotina*) and eastern cottonwood. Identified mammals included eastern cottontail (*Sylvilagus floridanus*) and whitetailed deer (*Odocoileus virginianus*). Identified birds included red-winged blackbirds (*Agelaius phoeniceus*), American robin (*Turdus migratorius*), northern cardinal (*Cardinalis cardinalis*), field sparrow (*Spizella pusilla*), domesticated pigeons (*Columba livia*), American coot (*Fulica americana*), common flicker (*Colaptes auratus*), American kestrel (*Falco sparverius*), and wild turkey (*Meleagris gallopavo*).

12.4.2 Threatened and Endangered Species

Endangered species are those organisms whose prospects for survival in an area are assumed to be in immediate danger because of a loss or change in habitat, over-exploitation, predation, competition, or disease. Threatened species are those who may become endangered if conditions surrounding the species begin, or continue to, deteriorate. There are two federally listed endangered species that can potentially be found at (or adjacent to) the Sites. Those species include the Indiana bat (*Myotis sodalis*) and pallid sturgeon (*Scaphirhynchus albus*). One federally listed threatened species that has been recorded in St. Clair County is the decurrent false aster (*Boltonia decurrens*). USEPA (1997b) noted that the decurrent false aster is found in alluvial prairie and marshland in river floodplains. It was concluded by USEPA (1997b) that it was unlikely that this species would be found at the Sites because of the history of extensive disturbance, though the USFWS has suggested that the habitat information is inaccurate.

A federally listed species that is known to winter in the region and identified in the area is the bald eagle (*Haliaeetus leucocephalus*). The bald eagle has been recently upgraded to threatened status from endangered by the U.S. Fish and Wildlife Service.

USEPA (1997b) did list several state-listed bird species that are likely to utilize the Sites. Those species include the black-crowned night heron (*Nycticorax nycticorax*), little blue heron (*Egretta caerulea*), snowy egret (*Egretta thula*), great egret (*Casmerodius albus*), and pied-billed grebe (*Podilymbus podiceps*). The great egret and pied-billed grebe are listed as threatened by the State of Illinois; the other three species are listed as endangered by the State. Only the black-crowned night heron has been sighted within two miles of the Sites.

Additionally, there are 18 federally or state (either Illinois or Missouri) listed fish species that have been historically shown to be present in the main stem of the Mississippi River in the region of the Sites (USGS, 1999). Those species include:

Alabama shad	*Alosa alabamae*	highfin carpsucker	*Carpiodes velifer*
alligator gar	*Atractosteus spatula*	Iowa darter	*Etheostoma exile*
bigeye shiner	*Notropis boops*	lake sturgeon	*Acipenser fulvescens*
blacknose shiner	*Notropis heterolepis*	mooneye	*Hiodon tergisus*
brown bullhead	*Ameiurus nebulosus*	northern pike	*Esox lucius*
central mudminnow	*Umbra limi*	pallid sturgeon	*Scaphirhynchus albus*
crystal darter	*Crystallaria asprella*	sicklefin chub	*Macrhybopsis meeki*
flathead chub	*Platygobio gracilis*	sturgeon chub	*Macrhybopsis gelida*
greater redhorse	*Moxostoma valenciennesi*	trout-perch	*Percopsis omiscomaycus*

12.4.3 Sensitive Habitats

Sensitive habitats include those ecological systems that support endangered or threatened species (either federally or state listed) or support wetlands. Given the lack of endangered or threatened species expected to be found on the Sites (USEPA, 1997b), habitat to support these species is not expected to be present. Menzie-Cura (1999) noted that a pair of bald eagles attempted to nest on the southern end of Arsenal Island, south of the Sites, in 1993. While the pair failed in their first attempt, it is not know whether later attempts were successful. A nest was observed by Menzie-Cura in 1996, but it did not appear to be in use.

The Clean Water Act defines wetlands as "those areas that are inundated or saturated by surface or groundwater at a frequency or duration sufficient to support, and that under normal circumstances does support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas" (33 CFR 230.3).

A review of the National Wetland Inventory (NWI) map for the Sites, prepared by the U.S. Fish and Wildlife Service, indicates that a substantial portion of the Sites P and Q have been categorized as wetlands. These wetlands are listed as palustrine wetlands, dominated by deciduous forests, shrub/scrub plant species, or emergent plant species. Palustrine wetlands are bounded by uplands or any other type of wetlands and may be situated shoreward of lakes, river channels or in floodplains (Cowardin et al., 1979). Shrubs are woody plant species that range from 3 to 20 feet in height. Emergent plants are those species in which at least a portion of the foliage and all of the reproductive structures extend above the surface of any standing water. Typical of this type of plant include cattails (*Typha* sp.), common reed (*Phragmites australis*), rushes (*Juncus* sp.) and sedges (*Carex* sp.). Emergents are usually found in shallow water or on saturated soils. Details of these wetlands will be developed during the field activities to be conducted in support of this ERA.

12.5 Conceptual Site Model

One of the most critical elements of the ERA scoping process is the development of the CSM. The CSM describes the hypothesized source of COPECs, routes of exposure and transport, and ecological receptors associated with the Sites. The CSM serves as the rationale for the development of sampling plans and protocols, the selection of assessment and measurement endpoints, and the identification of receptors of concern. The CSM can be revised, as new site-related information becomes available.

The following sections describe in greater detail the CSM for the aquatic and terrestrial pathways associated with the Sites.

12.5.1 Aquatic Pathways

The environmental fate of COPECs is determined by the cumulative interaction of transport and transformation processes (Paustenbach, 1987). Once released into the environment, the chemicals may partition among various media (e.g., soil, water, and air). The transport processes that define the movement of chemicals between compartments are highly dependent upon the physico-chemical properties of both the chemical and the environmental media, and thus, have a direct bearing on the potential risks to the exposed populations (Paustenbach, 1987). The ability of a chemical to proceed through a migration pathway and reach an exposed receptor is directly related to the chemical properties of the constituent and the physical characteristics of the pathway.

The primary aquatic pathway of concern with the Sites is the potential discharge of groundwater containing COPECs to the Mississippi River. Two of the five Sites are located in close proximity to the east bank of the Mississippi River (Sites Q and R). The other three Sites (Sites O, P, and S) are located 1500 to 2000 feet east of the riverbank. Solid and liquid industrial and municipal wastes were disposed at these facilities from the 1950s to the 1980s. At two of the disposal sites, wastes were placed in former borrow pit excavations (Sites Q and R). Wastes were placed in excavations at two other disposal sites (Sites O and S), however, these excavations were made solely for the purpose of waste disposal. Wastes were placed on grade at the fifth disposal site (Site P). It is likely that the excavations at Sites Q and R went to or below the water table to maximize the amount of borrow material. It is unlikely that the excavations for Sites O and S extended to the water table since these disposal sites needed only shallow excavations, 5 to 10 feet deep, to accommodate the materials placed in them.

Whether or not the waste disposal excavations extended to or below the water table, the potential exists for constituent migration through the groundwater system. The aquifer beneath Sauget Area 2 consists of three distinct hydrogeologic units: 1) the Upper Hydrogeologic Unit with fine-grained, silty sands, 2) the Middle Hydrogeologic Unit with clean, medium to coarse sand and 3) the Deep Hydrogeologic Unit with clean, medium to coarse sand and gravel.

Leachate migrating from the waste disposal areas could enter these hydrogeologic units and then discharge to the river via groundwater. The ultimate discharge point for these units is the Mississippi River.

If constituents of concern are transported through the groundwater system, they would be discharged into the Mississippi River. COPECs that are discharged through groundwater will first pass through the sediments of the river channel prior to entering the water column. In coarse-grained sediments with little organic material, the dissolved groundwater-borne COPECs will pass directly to the water column. In fine-grained or organic rich sediments, a portion of the groundwater-borne constituents may adhere to sediment particles. Whether the constituents remain in the sediment or are dissolved again will depend on its chemical characteristics. Those chemicals with high organic carbon partition coefficient (K_{oc}) values will have a greater affinity for sediment, especially sediment that is high in organic matter. Such constituents would tend to remain sorbed onto sediment particles and migration would occur as a result of sediment movement, not chemical movement.

The primary mechanisms by which chemicals migrate from sediments into the water column are through desorption from sediment particles, resuspension via physical disturbance and resuspension followed by food chain transport. COPECs that are dissolved in groundwater and adsorb onto sediment particles as groundwater wells up through the sediment base may desorb from the sediment particles over time, depending upon the K_{oc} value. For some high K_{oc} constituents, such as PCBs and chlorinated pesticides, desorption from sediment particles, especially those with a high organic content, is very slow, if not minimal. For other constituents, such as polycyclic aromatic hydrocarbons (PAHs) and organic solvents, desorption is much more rapid and can lead to a steady source of the constituent into the water column.

A physical disturbance of the sediment by anthropogenic activities like dredging and prop wash from boats, or natural activities such as flooding can cause resuspension of the chemicals within the sediment. This resuspension may be long- or short-term depending upon the size and solubility of the compound and the size of the sediment particle. Re-suspended particles to which these constituents are sorbed can be either organic matter or inorganic particulates. While in the water column, pelagic flora and fauna may be directly exposed to the re-suspended chemicals as they are transported downstream to other sites. These mobilized constituents in surface water may then be transported through the food chain to higher order trophic levels (i.e., piscivorous and omnivorous wildlife).

Once in the Mississippi River, the primary migration pathways for chemicals that are discharged from groundwater would be diffusion throughout the water body, and/or settling and bioaccumulation in the food chain. The process of diffusion is an ameliorating process because the compound is reduced in concentration. Constituents that are dissolved in groundwater and discharged into the river will pass through the sediment layer and be diffused by the larger volume of the receiving water body. Diffusion is further enhanced by the flow of water within the river upstream to downstream by increasing the rate of diffusion and moving the diluted chemicals out of the recharge zone. For species of limited mobility or sensitive life stages that may inhabit the discharge zone in close proximity to the discharge point for long periods of time, the potential for adverse impacts from exposure can still exist. However, in a river of extreme volume and flow, such as the Mississippi River in the vicinity of the Sites, dilution can be a major limiting factor to the potential for exposure of aquatic organisms to elevated levels of COPECs.

The settling of suspended particles or precipitation of dissolved chemicals removes the compound from the water column, but may increase the direct exposure to benthic organisms and bottom-rooting aquatic plants. Food chain transport from primary producers through the various trophic levels of consumers is compound specific and can lead to exposure of receptors that either live on the Sites or that come into contact with Site-related constituents that have been transported off-Site. However, food chain transport through the food web that is based on the benthic community includes a major contribution from detritus and not from primary production from rooted vegetation or epiphytic and planktonic plant species.

Another potential migration pathway to the Mississippi River is discharge via storm water runoff. Because Sites O, P, and S are found east of the flood control dike that runs parallel to the river, stormwater runoff would not be a concern. This pathway is not likely to be a major migration pathway at the Sites west of the flood control dike because the areas are covered with vegetation, soil caps, or impermeable pavement. Should it occur, surface water runoff would carry COPECs to the Mississippi River that are either dissolved in the water or adsorbed onto sediment particles. While runoff from Site R is likely limited due to the vegetation over the present cap, there are some areas of Q where runoff may occur. However, it is not presently know whether runoff areas or patterns are such that they drain surface soils containing COPECs. A runoff study will be conducted as part of the remedial investigation activities planned for Site Q. Once in the river, dissolved or suspended COPECs in surface water would be diluted and transported as described above.

Due to the ecological significance of the Mississippi River and the recreational and commercial importance of its natural resources, the aquatic pathway is the most significant pathway associated with the Sites. As shown in Figure 12-1 (Aquatic Conceptual Site Model for the Mississippi River), constituents that are possibly released in groundwater can migrate through the sediments into the water column in the vicinity of the discharge point. The sampling that will support this ERA, described in accompanying FSPs and QAPPs (Volumes 3A and 3B), is designed to investigate the magnitude of this potential release by examining the surface water and sediment in the likely discharge areas for groundwater. The biological sampling included in the field studies is intended to provide data on the conditions of vertebrates and invertebrates associated with the aquatic system for the purpose of evaluating the potential ecological effects resulting from a possible groundwater discharge.

As noted in Section 12.4.1, a number of sources of constituents similar to those found in the Mississippi River adjacent to the Sites and in on-Site areas, including agricultural runoff, wastewater treatment facilities, industrial discharges, and non-point pollution sources, exist upstream of the Sauget Sites. With the high velocity and large volume of water in the river in the vicinity of the Sites, rapid downstream transport (i.e., toward the Sites) of materials from these upstream sources is expected. It will be necessary to use care when evaluating the potential for adverse ecological effects and identifying those that are Site-related versus those that are not. To account for the potential effects of these upstream sources and the effects of high velocity, appropriate reference locations will be utilized to place any suspected effects in the context of regional conditions and to serve as a comparison for the determination of Site-related effects.

The aquatic pathways described in this section also apply to the ponded areas located in the southern portion of Site Q. COPECs that are present in the surrounding soils may migrate into the ponds, thereby exposing biota that use or live in the ponds. As previously mentioned, the ponds were drained and have recently refilled as a result of stormwater influence. As such, the aquatic community that is present is extremely limited and probably restricted to early successional aquatic plants and some early colonizing benthic invertebrates. However the ponds may serve as a water source for terrestrial vertebrates and a breeding spot for amphibians or aquatic birds. A conceptual CSM for the ponded areas within Site Q is presented as Figure 12-3.

12.5.2 Terrestrial Pathways

The migration of COPECs within soils may result in either direct exposure through contact with the soil or indirect exposure through the food chain to faunal communities supported by the available habitat at the Sites. Biota may come in direct contact with chemicals in soil while foraging and/or burrowing. The vectors by which chemicals in the soil may potentially be introduced into biota are direct ingestion (primary source), dermal absorption, or inhalation. The USEPA has determined that inhalation comprises less than 0.1% of total exposure and direct contact comprises approximately 1 to 11% of total exposure (USEPA, 2000). Indirect exposure occurs when a COPEC is assimilated by a species (e.g., prey/food item) at one trophic level, bioaccumulated by that trophic level, and transferred to the next trophic level through consumption.

Chemicals in surface soils may potentially move from the soils up the food chain through bioaccumulation and biomagnification processes. Soil dwelling organisms (i.e., soil invertebrates and small mammals) may bioaccumulate chemicals from direct and incidental ingestion and through direct contact with the soil.

The terrestrial pathways at the Sites are associated with the five Sites. Site O includes four lagoons that were capped with two feet of clay in 1980. Between 1966/67 and 1978, these lagoons were used to dispose of clarifier sludge from the Sauget Physical Chemical Wastewater Treatment Plant (a publicly owned treatment works). During its operation the treatment plant and associated lagoons received and treated industrial and municipal wastewater. It has been reported that approximately 10 million gallons per day of wastewater was treated, more than 95% of which was from area industries.

Site P was operated as an IEPA-permitted landfill from 1973 to approximately 1980, accepting general wastes, including diatomaceous earth filter cake and non-chemical wastes. Site P is currently inactive and partially covered by parking areas; however, access to this part of the Sites is not restricted.

Site Q is a former subsurface and surface disposal area that accepted various wastes including municipal waste, liquid chemical wastes, septic tank pumpings, drums, organic and inorganic wastes, solvents, pesticides and paint sludges. It also took plant trash, waste from other industrial facilities and demolition debris. Most of Site Q is covered with highly permeable black cinders. Site S was a disposal area. The northern portion of this part of the Site is grassed and its southern portion is covered with gravel and fenced.

Site R is a closed industrial-waste disposal area that accepted hazardous and non-hazardous bulk liquid and solid chemical wastes and drummed chemical wastes that included organics, inorganics, solvents, pesticides, and metals. The Site is capped with a clay cover whose thickness ranges from 2 feet to approximately 8 feet.

As shown in the Terrestrial Conceptual Site Models, terrestrial receptors may be exposed to constituents located in these Sites. A CSM for the terrestrial portion of the Sites is presented as Figures 12-2. If constituents are present in surface soils, the migration pathways described above could lead to the possible exposure of plant and wildlife receptors to Site-related constituents. Surface soil sampling in each area is intended to characterize the constituents present in soil in areas frequented by biological receptors. The biological sampling is intended to provide data to support an assessment of the potential for adverse ecological effects from the presence of these constituents.

Terrestrial receptors may also be exposed to COPECs present in air and in surface water. Both of these pathways are considered to be minor. Compounds with a high degree of volatilization (e.g., volatile organic compounds) can expose mammals to vapors that produce ecological effects. However, noting the age of the Sites, the partial coverage by caps and impermeable layers, and the level of historic disturbance that would act to accelerate the volatilization process, this pathway is considered to be insignificant (Sample and Suter, 1994).

Terrestrial receptors may be exposed to COPECs in water and sediments. Most small mammals (particularly rodents) obtain their water through ingestion of plants with high water content, rain puddles collected on the ground or on impervious surfaces, and from dew (Vaughn, 1986). Larger carnivores may drink from ponds, rivers, puddles, or lakes. Terrestrial receptors may also be exposed to COPECs in surface water (and sediment) through either incidental ingestion or through direct contact. Birds may be exposed through the accumulation of soil and sediment by use of these materials as grit.

12.6 Selection Of Chemicals Of Potential Ecological Concern (COPECS)

The QAPP/FSP lists target analytes for the Sites. These target analytes include volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), pesticides, herbicides, polychlorinated biphenyls (PCBs), dioxin/furans, and metals. The list of analytes has been developed in discussions with the USEPA.

The results of the proposed sampling and analysis of surface water, sediments, surface soils, and biota, will be used to select the list of COPECs. The number of proposed samples and the sample locations, in both the terrestrial and aquatic environments has been selected in discussions with the USEPA and are designed to reflect worst-case conditions in the two ecosystems.

Constituents will be retained on the list of COPECs and carried through the screening process if they exhibit any of the following characteristics:

- Toxic – Produce a harmful effect (is toxic), based on the scientific literature or direct measures of toxicity, to the receptors likely to inhabit the Mississippi River in the vicinity of the Sites and the terrestrial environment found within the five Sites.

- Bioaccumulative – Likely to bioconcentrate or biomagnify in the aquatic and terrestrial food chains likely to be at, or adjacent to, the Sites (determined as a constituents with an octanol-water partition coefficient (K_{ow}) greater than 1,000 [*Great Lakes Water Quality Initiative* (GLWQI) (40 CFR 122 et al., 1995)]; and

- Persistent – Likely to remain in environmental media over time frames that are long, relative to the life spans or exposure periods of receptors likely to inhabit the Mississippi River in the vicinity of the Sites and the terrestrial environment found within the five Sites.

For those constituents that are shown to be potentially toxic, bioaccumulative or persistent, additional screening parameters will be utilized to refine the list of COPECs. The criteria for final selection and evaluation include:

- Comparison to Background – The ERA will eliminate a constituent that occurs below the maximum concentration measured at a local reference area for a given medium.

- Frequency of Detection – The ERA will eliminate a constituent for evaluation if the constituent is detected in less that 5% of samples from a particular medium.

Ecotoxicological profiles for the COPECs will be included within the ERA.

12.7 *Identification of Receptors and Endpoints*

The analysis portion of the ERA includes the estimation of potential exposures of biological receptors to Site-related COPECs and the determination of the potential effects associated with those exposures. The assessment of effects is the determination of the relationship between the concentrations of COPECs potentially identified in various matrices at the Sites and the responses of ecological receptors to these concentrations. Exposure to ecological receptors will occur either directly through mechanisms such as ingestion, incidental contact, or inhalation, or indirectly through the consumption of prey species containing elevated concentrations of COPECs. Indirect or food chain exposure can potentially result in unacceptable risks to higher trophic level organisms without their being in close proximity to the Sites. This section outlines those components that will be utilized in the assessment of the potential ecological effects associated with the exposure of ecological receptors to COPECs at the Sites.

12.7.1 Identification of Receptors of Interest

The first step in the assessment of effects is the identification of those receptors of interest (ROIs) that will be evaluated in the ERA. As it is not feasible to evaluate the relationship of all potential chemicals of interest to every species at the Sites, ROIs have been selected to represent the organisms that might be present at the Sites most often or are likely to be most sensitive to the effects of the COPECs. Selection criteria for aquatic ROIs include the following factors specified in USEPA guidance (1989a, 1992, 1994, 1997a, 1998): (1) the occurrence of potentially complete pathways for exposure of ecological resources to chemicals in environmental media; (2) resident communities or species exposed to the highest concentrations of chemicals in environmental media; (3) species or functional groups considered to be essential to, or indicative of, the normal functioning of the affected habitat; and (4) the feasibility of completing a quantitative assessment for the identified pathways and receptors.

Species were selected as ROIs for this assessment based on the following criteria:

- Relative abundance and ecological importance within the identified habitats;
- Availability and quality of appropriate ecotoxicological research;
- The sensitivity of the organism to the COPECs;
- Importance of the trophic level in the ecosystem;
- The relative mobility and type of feeding habits; and
- The ability to bioaccumulate COPECs.

The following ROIs have been selected for use in preparing this ERA:

For the Mississippi River aquatic community:

- The benthic macroinvertebrate community;
- Local fin fish;
- Mink (*Mustela vison*); and
- Osprey (*Pandion haliaetus*).

Benthic invertebrates were selected as ROIs because they have the greatest exposure to bottom sediments that potentially contain COPECs and they are an important link in the aquatic food chain as a food source for bottom feeding fish species in the river. The use of macroinvertebrates as biological indicators of pollution in freshwater ecosystems has distinct advantages over purely physical and/or chemical analyses (Hellawell, 1977). Physical and chemical analyses reflect the characteristics of a waterbody during a single point-in-time; whereas, biomonitoring reflects environmentally induced changes that occur over a long period of time (sometimes a year or more). Therefore, if a constituent happens to be either at extremely low levels or absent at the time of physical and chemical analysis, a false reading as to the environmental quality of an ecosystem can be obtained.

Secondly, a broad range of constituents may affect an ecosystem, some of which may not have been identified yet. Relying solely on the physical and chemical analyses to evaluate potential ecological effects may lead to missing an important ecological effect. However, biological measures can be used to ensure that such effects are identified. Though care needs to be taken when interpreting the results of biological investigations. For example, measures of abundance are more relevant than measures of community structure when evaluating benthic communities, because abundance (not diversity) is more closely tied to the most important role of the benthic community (which is serving as a prey base for higher trophic level receptors). The absence of a sensitive indicator species may not be an indication that the benthic community as a whole is not performing its function as a food source for fish and other vertebrates. Similarly, effects to biological indicators indicate that further analysis needs to be performed to determine the exact nature of the stressor (e.g., physical disturbance, specific chemical stressor, etc.).

Local finfish were selected as ROIs because they are the dominant organisms in the water column and they may be exposed to COPECs in sediments and surface water. Fish represent a food source to higher order predators (both aquatic and semi-aquatic) and are important for both recreational and commercial fisheries.

The mink and osprey were selected as upper trophic level ROIs because the biological success of local populations of these organisms can be tied to the environmental health of supporting habitats. Both species are either found in the area, or have the potential for being found in the area. They all feed on fish, so they can be tied via the food web to the sediments and surface water of the Mississippi River. Additionally, both species are sensitive to constituents that biomagnify up the food web.

For the riverbank/floodplain terrestrial community:

- Prairie vole (*Microtus ochrogaster);*
- Short-tail shrew (*Blarina brevicauda*); and
- Red fox (*Vulpes fulva).*

The prairie vole was selected as a ROI because it is likely to be the dominant herbivore within the habitat provided by the Sites and with its small home range, could likely spend its entire life span within a Site. Shrews were chosen as an ROI because a large portion of their diet consists of earthworms that live within the soils of the Sites. The red fox is an upper trophic level carnivore potentially feeding on either shrews or other small rodents within the Sites.

For the riverbank/floodplain aquatic community within site-related ponds (only if water and aquatic organisms are present in the ponds at the time of the ERA supporting field activities):

- The benthic macroinvertebrate community, and
- Finfish or amphibians.

As discussed in Section 12.4.1, during a brief reconnaissance of Site Q conducted in January 2001, it was observed that the ponds did not contain standing water. If the ponds have refilled by the time the field studies to support this ERA are conducted, the benthic invertebrate community will be examined to assess the potential for the reestablishment of the benthic food base. However, even if the ponds have refilled with water, it is will not be likely that a finfish population will have been reestablished. The reestablishment of a finfish population will not likely occur until such time as a flood event overflows the bank of the Mississippi River and stocks the pond. However, the possibility for future impacts to fish will be examined through the evaluation of concentrations of COPECs that are identified in surface water.

While the ponds may not at present support a finfish population, they may support a breeding population of amphibians. Historic information (USEPA, 1997b) indicates that frogs were identified in the vicinity of the ponds. As a steady source of water is required for most amphibians to breed, COPECs that are found in surface water have the potential to impact amphibians during a sensitive point in their life cycle. Therefore, if no finfish are present, concentrations of COPECs in surface water will be evaluated for their potential to impact amphibians.

12.7.2 Assessment and Measurement Endpoints

The next step in the ecological risk assessment process is the identification of those endpoints that will be utilized in the ERA to evaluate the ecological effects associated with the potential exposure of ROIs to COPECs. Assessment endpoints are statements of the characteristics or attributes of the environment that are to be protected. This ERA will evaluate the following assessment endpoints:

- Assessment Endpoint #1: Evaluate the potential for adverse changes in the survival, reproduction, and growth of finfish populations utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in sediments, surface waters, and/or prey;

- Assessment Endpoint #2: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of piscivorous wildlife utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in prey;

- Assessment Endpoint #3: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of vermivorous wildlife utilizing the five Sites resulting from exposures to COPECs in prey;

- Assessment Endpoint #4: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of herbivorous wildlife utilizing the five Sites resulting from exposures to COPECs in soils and/or prey;

- Assessment Endpoint #5: Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of carnivorous wildlife utilizing the five Sites resulting from exposures to COPECs in prey; and

- Assessment Endpoint #6: Evaluate the potential for adverse changes in the survival, reproduction, and growth of potential finfish populations within the two ponds, or the potential for adverse changes in the reproductive potential of amphibian populations breeding within the two ponds located in Site Q resulting from exposures to COPECs in surface waters.

The assessment will evaluate ecological risks relative to these assessment endpoints in the Mississippi River and the five Sites. Data to evaluate these endpoints will be collected during field studies as specified in QAPPs and FSPs that have been prepared separately.

Since the above assessment endpoints generally cannot be measured directly, measurement endpoints have been identified. There are four types of measurement endpoints or lines of evidence that will be used to assess the status and potential changes in the attributes of the environment. The lines of evidence are:

1) Determination of the potential for ecological effects by the comparison of COPEC concentrations to media-specific *ecotoxicological benchmarks* derived from the literature;
2) Biological survey data of various ROIs which are direct estimates of the assessment endpoint;
3) Bioassays which are direct measures of the relative toxicity of constituents in a particular matrix; and
4) Estimation of potential for ecological effects from estimated exposures of higher trophic level organisms to COPECs based on food chain modeling.

A weight-of-evidence approach will be utilized in evaluating data collected for each of the measurement endpoints. Each line of evidence used in the weight-of evidence approach will be correlated in an exposure-response relationship in the attempt to demonstrate a relationship between the magnitude of exposure and the magnitude of effects. A weight-of-evidence approach (as detailed in Section 12.8) weighs each of the measurement endpoints by considering:

- The strength of association between the measurement endpoints and the assessment endpoints;
- Data quality; and
- Study design and execution.

The strength of association refers to how well a measurement endpoint represents an assessment endpoint. The greater the correlation between the measurement and assessment endpoints, the greater the weight given to that measure of effect in the risk analysis.

The weight assigned a measurement endpoint also depends on the quality of the data as well as the overall study design and execution. The FSPs and QAPPs describe a sampling program that will provide information to evaluate each measurement endpoint. However, the ERA must evaluate the sampling effort and variability and uncertainties associated with the results following implementation. *The risk characterization gives higher weight to measurement* endpoints that are based on good quality data and are obtained using study designs that account for confounding variables.

Considerable uncertainty is associated with estimating potential ecological risks, because ecological systems are complex and exhibit high natural variability. Measurement endpoints typically have specific strengths and weaknesses related to the factors discussed above. Therefore, it is common practice to use more than one measurement endpoint to evaluate each assessment endpoint. Measurement endpoints are as follows:

> **Assessment Endpoint #1: Evaluate the potential for adverse changes in the survival, reproduction, and growth of finfish populations utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in sediments, surface waters, and/or prey.**

Measurement Endpoints for Assessment Endpoint #1

a. The first line of evidence will be the evaluation of COPEC data obtained through the chemical analysis of surface water. The ability of surface water to potentially support viable populations of finfish will be assessed by evaluating concentrations of COPECs identified in surface water samples collected upstream, downstream and in the vicinity of the Sites. This evaluation will be conducted by comparing identified concentrations of COPECs from all sampling locations to relevant ecologically-based threshold benchmarks, including the State and Federal Ambient Water Quality Criteria for the protection of fish. The comparison will be established in terms of an exposure-response gradient both horizontally and vertically across all sampling locations. The resulting relationship will be taken as a line of evidence in conjunction with the bioassay and fish tissue data in support of an exposure-response relationship.
b. The second line of evidence will be the evaluation of surface water bioassay data. The ability of surface water to potentially support viable populations of finfish will be assessed by evaluating by survival rate test data from bioassays conducted on surface water samples collected upstream, downstream and in the vicinity of the Sites. The results of the bioassays will be examined in terms of an exposure-response relationship in correlation with the surface water chemical data and the fish tissue data. The resulting relationship will be taken as one line of evidence in conjunction with the chemical and fish tissue data in support of an exposure-response relationship.
c. The third line of evidence is the evaluation of whole body COPEC concentrations identified in fish collected upstream, downstream and in the vicinity of the Sites. The ability of surface water to potentially support viable populations of finfish will be assessed through the evaluation of concentrations of COPECs identified on a whole body basis in selected fish species. The body burden levels will be compared to tissue residue data from the literature (e.g., Jarvinen and Ankely, 1999) that indicates potential ecological concerns. The resulting relationship will be taken as one line of evidence in conjunction with the chemical and bioassay data in support of an exposure-response relationship.

d. Another line of evidence will be the evaluation of COPEC data obtained through the chemical analysis of sediments. The ability of the benthic community to perform its role as a prey base for finfish will be evaluated by comparing the concentrations of COPECs in sediments collected in upstream, downstream and Site-related sampling locations to appropriate sediment quality benchmarks for the protection of benthic macroinvertebrates. The comparison will be established in terms of an exposure-response gradient vertically across all sampling locations. The resulting relationship will be taken as one line of evidence in conjunction with the bioassay test data in support of an exposure-response relationship.

e. Another line of evidence will be the evaluation of sediment bioassay data. The ability of the benthic community to potentially perform its role as a prey base for finfish will be evaluated by evaluating the survival rates of bioassay test organisms following exposure to sediment samples collected from upstream, downstream, and Site-related sampling locations. The evaluation of the survival rates will be established in terms of an exposure-response gradient horizontally across all sampling locations. The resulting relationship will be taken as one line of evidence in conjunction with the chemical test data in support of an exposure-response relationship; and

f. A final line of evidence will be the evaluation of measures of benthic community function.

Rationale for Measurement Endpoints 1a through 1i

Because of the ecological, recreational and commercial importance of fish populations in the Mississippi River, the release of Site-related COPECs at sufficient concentrations could have an ecological impact to these resources. As described above, to evaluate this potential effect, a number of lines of evidence will be examined.

The measurement endpoints identified above were selected to evaluate the potential pathways that would result in the exposure of fish to Site-related COPECs. Those endpoints associated with surface water measurements are intended to provide an assessment of the ability of that matrix to potentially support fish. As the sampling areas are fairly limited in comparison to the size of the Study Area, assumptions will be made based on best professional judgment regarding the exposure-response relationship for upstream to downstream as well as crossstream relationships. The most critical of these measurements, the surface water bioassays, is intended to identify whether surface water in Site-related areas significantly affects the survival of finfish. If the bioassays indicate that the survival of test species is impaired when exposed to Site-related water, then correlation with the chemical analyses will assist in the development of exposure-response relationship and will help identify which COPEC potentially may be producing the toxicity.

While the evaluation of surface water will provide an estimate of the ability of fish to exist in Site-related waters, the analyses of fish tissue will provide a direct measurement of the potential for ecotoxicological impacts from exposure of fish to COPECs. The measure of concentrations of COPECs within fish tissue will be a direct measure of the assimilative and bioaccumulative capacity of COPECs that may be identified in surface water and will be a measure of the potential toxicity of those COPECs to fish. Care will be used when interpreting these data because discriminating between Site-related and non-Site-related causes is difficult in wide ranging species such as fish. Additionally, measurement of COPECs within fish tissue will allow for the assessment of the potential exposure and possible effects to higher trophic level organisms that feed on fish.

The evaluation of the benthic macroinvertebrate community provides a measurement of its ability to function as a prey base for finfish. By examining the benthic community through multiple measurement endpoints, as identified in the sediment triad approach, the ERA will be able to correlate chemical concentrations in sediments with measures of toxicity and biological integrity, while comparing the measured concentrations to literature values of possible effects. The benthic community assessment, the most critical measure, will provide evidence of the ability of macroinvertebrates to live in sediment found in Site-related areas. The chemical analyses will help identify which of the COPECs may be responsible for any observed toxicity. This approach will be further enhanced by the inclusion of bioaccumulation studies that examine the potential for migration of chemicals bound within the sediments to biota living within it.

<u>Assessment Endpoint #2:</u> **Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of piscivorous species utilizing the Mississippi River in the vicinity of the Sites resulting from exposures to COPECs in prey.**

Measurements Endpoints for Assessment Endpoint #2

a. Potential risks to mink will be estimated by comparing an estimated average daily dose for each potential COPEC to a toxicity reference value for each potential COPEC identified in the literature. Exposure concentrations to mink will be estimated using finfish COPEC body burdens and a food chain model; and
b. Potential risks to ospreys will be estimated by comparing an estimated average daily dose for each potential COPEC to a toxicity reference value for each potential COPEC identified in the literature. Exposure concentrations to ospreys will be estimated using finfish COPEC body burdens and a food chain model described in Section 12.7.6.

Rational for Measurement Endpoints 2a through 2c

While COPECs that have been identified in fish may not produce direct ecotoxicological effects in fish, if the constituents are bioaccumulative, they may have the potential for producing effects in higher trophic level organisms that feed on fish. This may result in indirect impacts to more wide-ranging species, to species that are especially sensitive to particular COPECs, or to species that have been assigned special status because of low population levels or habitat requirements. For that reason, a food chain model will be employed to assess the potential exposure of two piscivorous wildlife species to COPECs in fish tissue. Using COPECs identified in small forage fish (four inches to ten inches in length), the potential for ecological risks to mink feeding along the Mississippi River and in the on-Site ponds (if there are fish present) will be calculated. The potential for ecological risks to osprey feeding along the Mississippi River will be calculated using COPECs identified in large forage fish (six inches to fourteen inches). For each species, the Average Daily Dose (ADD) of COPECs will be compared to toxicity reference values (TRVs) for that species identified from the literature. If the ADDs exceed the TRVs in a large number of locations, then the potential exists for adverse effects.

<u>Assessment Endpoint #3:</u> Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of vermivorous wildlife utilizing the five Sites resulting from exposures to COPECs in prey.

<u>Measurements Endpoints for Assessment Endpoint #3</u>

a. Potential risks to short-tail shrews will be estimated by comparing an estimated daily dose for each potential COPEC to a TRV for each potential COPEC identified in the literature. Exposure concentrations to short-tail shrews will be estimated using earthworm COPEC body burdens and a food chain model described in Section 12.7.6.

Rational for Measurement Endpoint 3a

Vermivorous wildlife have been identified by the USEPA as a trophic level of concern at the five Sites. Because the 1/3 of the diet of these animals is earthworms, which live in contact with soils containing COPECs and can potentially accumulate COPECs, potential ecological risks to vermivorous wildlife will be evaluated by using COPEC earthworm tissue residue data to estimate ADDs for short-tail shrews. Earthworm residue data will be collected by the use of earthworm bioassay tests using soil collected from the five Sites. ADDs will be compared to TRVs and if the ADDs exceed the TRVs in a large number of locations, then the potential may exist for adverse effects to short-tailed shrews.

<u>Assessment Endpoint #4:</u> Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of herbivorous wildlife utilizing the five Sites resulting from exposures to COPECs in soils and/or vegetation.

<u>Measurements Endpoints for Assessment Endpoint #4</u>

a. The ability of the plant community to provide habitat for herbivorous wildlife will be measured by the comparison of concentrations of COPECs in surface soils at the Sites to appropriate surface soil quality benchmarks for the protection of plants; and

b. Potential risks to prairie voles will be estimated by comparing an estimated daily dose of each potential COPEC to a TRV for each potential COPEC identified in the literature. Exposure concentrations to prairie voles will be estimated using plant COPEC tissue concentrations and a food chain model described in Section 12.7.6.

Rational for Measurement Endpoints 4a through 4c

If soil contains COPECs, plant roots may be in constant contact with, or at least in close proximity to, constituents that can be either toxic to plants or that can be translocated up into the various parts of a plant and then consumed by higher order trophic level receptors. Cell membrane barriers within roots act to restrict the movement of COPECs into the root cortex; thereby, limiting the translocation and the actual expression of ecological effects. However, there is the potential that certain COPECs may accumulate in various plant tissues and then be consumed by herbivorous wildlife. The collection of plant tissue for chemical analysis will provide a direct measurement of the uptake potential of COPECs by plants and will allow for the assessment of impacts to higher trophic level organisms. Ecological risks to herbivorous wildlife will be estimated through the determination of ADDs for a prairie vole and comparison of the ADD to the TRV derived from the literature. If the ADD exceeds the TRV at a large number of locations, then there is the potential for adverse ecological risks.

Screening of COPECs against soil benchmarks allows for the estimation of potential ecological impacts to plants at the five Sites. However, this is a highly conservative assessment as the development of soil screening benchmarks is still in its infancy, and the existing data are often dependent upon weak correlations, laboratory conditions, and extreme uncertainty factors.

<u>Assessment Endpoint #5:</u> Evaluate the potential for adverse changes in the survival, reproduction, and growth of populations of carnivorous wildlife utilizing the five Sites resulting from exposures to COPECs in prey.

<u>Measurement Endpoints for Assessment Endpoint #5</u>

Potential risks to the red fox will be estimated by comparing estimated daily dose of COPECs to TRVs identified in the literature. Exposure concentrations to red fox will be estimated using plant COPEC tissue concentrations, earthworm COPEC tissue residue concentrations, and a food chain model described in Section 12.7.6 that estimate COPEC body burdens in prairie voles and shrews.

Rational for Measurement Endpoints 5a

Estimated concentrations of COPECs in the prairie vole and the short-tail shrew may not produce direct ecotoxicological effects. However, if the constituents are bioaccumulative, they may have the potential for producing effects at higher trophic levels. This may result in indirect impacts to more wide-ranging species, to species that are especially sensitive to particular COPECs, or to species that have special status because of population levels or habitat requirements. For that reason, a food chain model will be employed to assess the potential exposure of an upper level carnivore (the red fox) to COPECs in voles and shrews. Using estimated concentrations of COPECs in prairie voles and short-tail shrews, the potential for ecological risks to red fox feeding within the five Sites will be calculated. The Average Daily Dose (ADD) of COPECs will be calculated and compared to toxicity reference values (TRVs) for the red fox. If the ADD exceeds the TRV in a large number of locations, then the potential exists for adverse effects.

Assessment Endpoint #6: Evaluate the potential for adverse changes in the survival, reproduction, and growth of potential finfish populations within the two ponds, or the potential for adverse changes in the reproductive potential of amphibian populations breeding within the two ponds located in Site Q resulting from exposures to COPECs in surface waters.

Measurement Endpoints for Assessment Endpoint #6

As previously indicated, in January 2001 the two ponds located in Site Q did not contain standing water, though more recent information provided by the USEPA indicates that the ponds have refilled. Information from previous studies (USEPA, 1997b) indicates that the ponds contained water and did support a standing population of fish. While the ponds may be filled during the ERA supporting field activities, it is highly unlikely that they would have developed a finfish community by that time. Therefore, should the ponds be filled at the time of the ERA supporting field activities and contain a finfish population, then the same measurement endpoints used for evaluating the fish population in the Mississippi River (measurement endpoints 1a through 1i) will be used for evaluating the viability of a potential aquatic community within the ponds. However, the pond could provide breeding habitat for amphibians (chorus frogs have been historically identified in the vicinity of the ponds). If a finfish community is not present at the time of the ERA supporting field activities, then concentrations of each potential COPEC that may be identified in surface water will be compared to TRVs for its associated COPEC that have been identified in the literature to evaluate the potential for ecological risks to breeding amphibians. Bioassay data will also provide information on the likelihood of COPECs in surface water to impact the ability of amphibians to breed within the ponds.

Methodologies for collecting the data required to measure the effects are described in separate FSPs and QAPPs for both the aquatic and terrestrial portions of the Sites.

12.7.3 Ecotoxicological Benchmarks

As described in Section 12.7.2, some of the measurement endpoints to be used in the assessment of potential risks to ecological receptors in the terrestrial and aquatic environments include the comparison of ecotoxicological benchmarks to Site-related data for various media. These benchmarks are risk-based screening concentrations that will be used to evaluate the concentrations of chemicals detected in surface water, sediment, and surface soil in aquatic and terrestrial areas of interest at, and adjacent to, the Sites. They are species-specific and chemical-specific, and will represent chemical concentrations in a matrix, below which adverse effects will not likely occur.

However, the benchmarks are not intended to serve as reference levels that will trigger specific actions if exceeded. The exceedance of a benchmark is not confirmation that an ecological impact is occurring. Rather, the benchmarks are primarily intended to help focus and prioritize project objectives and data requirements during the planning and implementing of site-specific investigations, by identifying constituents and particular areas of sites that may pose potential risks to aquatic and terrestrial ecological receptors.

12.7.3.1 Surface Water Screening Benchmarks

Concentrations of COPECs identified in surface water will be compared to the National Ambient Water Quality Criteria (NAWQC) (USEPA, 1996). Acute and chronic NAWQC values were developed for the protection of aquatic life in freshwater environments. Acute and chronic Tier II Secondary Values (SVs) will also be used during the screening process when a NAWQC is unavailable. Together, these criteria or benchmarks provide an initial basis for evaluating potential impacts to ecological receptors. The surface water screening benchmarks are described below.

<u>National Ambient Water Quality Criteria and Tier II Secondary Values</u>

Acute NAWQC values are calculated as ½ the final acute value, which is the fifth percentile of the distribution of 48- to 96-hour LC_{50} values or equivalent EC_{50} values for each criterion chemical. Acute values correspond to concentrations that would cause less than 50% mortality in 5% of the exposed population in a brief exposure. Chronic NAWQC values are the final acute value divided by the final acute:chronic ratio (Suter, 1996). NAWQCs are available for a limited number of compounds.

In the absence of NAWQCs, Tier II SVs will be calculated using the methodology presented in the *Great Lakes Water Quality Initiative* (GLWQI) (40 CFR 122 et al., 1995). Tier II acute and chronic SVs are based upon fewer data than are required for the calculation of NAWQCs and safety factors to account for the lack of complete data. The use of safety factors is designed to result in concentrations that are expected to be lower (i.e., more stringent) in approximately 80% of the cases than the NAWQC for a chemical calculated with sufficient test data (Suter, 1996). NAWQCs for the protection of aquatic life are based on thresholds for statistically significant effects on individual responses of fish and aquatic invertebrates. Those thresholds correspond to approximately 25% reductions in the parameters (i.e., survival, growth or reproduction) of chronic fish tests (Suter et al., 1987). Because of the compounding individual responses across life stages, the chronic NAWQCs frequently correspond to much more than 20% effects on a continuously exposed fish population (Barnthouse et al., 1990). Therefore, an exceedance of the chronic Tier II SV is assumed to correspond to a 20% or greater effect (i.e., reduction) on the survival, growth, or fecundity of the fish community. The acute and chronic Tier II SVs used in this assessment and their derivation are described in detail in Suter and Mabrey (1994) and Suter (1996).

For compounds including the dioxins and furans, toxicity equivalent factors (TEFs) have been developed for fish (Van den Berg et al., 1998.) that indicate the relative potency of the individual 2,3,7,8-substituted congeners to the toxicity of 2,3,7,8-tetrachloro-dibenzo-p-dioxin (2,3,7,8-TCDD). These TEFs will be used to evaluate surface water concentrations of the dioxin and furan congeners.

To assess the potential for ecotoxicological impacts to amphibians breeding in the ponds, concentrations of COPECs in the surface water of the ponds will be compared to relevant benchmarks. The scientific literature will be evaluated to determine appropriate benchmarks for amphibians. If none are available out of the literature, then AWQC values will be used instead.

12.7.3.2 Sediment Screening Benchmarks

Ontario Lowest Effects Levels (LELs) and Severe Effects Levels (SELs) (Persaud et al., 1993) will be used for screening sediments. If LEL and SEL values are unavailable for a chemical constituent, a hierarchical approach to identifying other sediment benchmarks will be used. If LEL and SEL values are not available, then the order in which other sources will be considered in the identification of a benchmark for a particular constituent include:

1. Threshold Effect Level (TEL) and Probable Effects Level (PEL) values developed by Smith et al. (1996)
2. USEPA Sediment Quality Benchmarks or Criteria (SQBs and SQCs, respectively) (USEPA, 1998b); and
3. National Oceanic and Atmospheric Administration (NOAA) Effects Range-Low (ER-L) and Effects Range-Median (ER-M) values (Long et al., 1995).

These values will be used for screening purposes. An exceedance of these screening benchmarks does not necessarily indicate that the benthic community has been adversely affected.

Lowest Effects Levels and Severe Effects Levels

The Ontario Ministry of the Environment (MOE) has prepared provincial sediment quality guidelines (SQGs) using the Screening Level Concentration (SLC) Approach. The SLC approach estimates the highest concentration of a particular constituent in sediment that can be tolerated by approximately 95% of benthic infauna (Neff et al., 1988). The SLC is derived from synoptic data on sediment chemical concentrations and benthic invertebrate distributions. These values are based on Ontario sediments and benthic species from a wide range of geographical areas within the province (Persaud et al., 1990). The guidelines define levels of ecotoxic effects and are based on the chronic, long-term effects of constituents on benthic organisms (Persaud et al., 1993).

The SELs for organic constituents will be normalized for the site-specific total organic content within the sediment at each location. Concentrations of organic compounds detected in sediment at each location will be compared to the TOC-normalized SEL to determine the magnitude or probability of the potential for an ecological impact on the benthic macroinvertebrate community at that location.

LELs and SELs are screening values used for the identification of potential ecological impacts in sediments. They do not take into account site-specific attributes such as bioavailability, bioaccumulation, or the acclimation of organisms to the presence of a COPEC. Therefore, they must be used in conjunction with other sediment screening tools such as field observations and bioassays.

Threshold Effect Level and Probable Effect Level Values

If MOE SQGs are not available for a particular constituent, then TEL and PEL values will be utilized for comparison of sediment concentrations. TEL and PEL values were developed based on a review of sediment chemistry and bioassay data stored in the Biological Effects Data Base for Sediments (BEDS), along with parameters that can affect bioavailability. The TEL values are based on the geometric mean of the 15th percentile concentration from the *effect* data set and the 50th percentile from the *no effect* data set. The PEL is calculated from the *effect* data set and the 85th percentile concentration of the *no effect* data set.

Sediment Quality Criteria and Sediment Quality Benchmarks

If MOE SQGs and TEL/PEL values are not available, then USEPA SQCs and SQBs (USEPA, 1998) values will be utilized for comparison of sediment concentrations. The USEPA SQCs and SQBs were derived by the equilibrium partitioning (EqP) method that uses the K_{ow} and determines the sorption capacity of the sediment by the mass fraction of organic carbon in the sediment (USEPA, 1998b). The SQCs and SQBs are based on the toxicity of compounds in water expressed as the NAWQC or Tier II Secondary Chronic Values (SCVs) and partitioning of the constituent between sediment organic matter and pore water. These benchmarks are calculated using the site-specific TOC content in the sediments at each location. If USEPA SQBs or SQCs are unavailable, SQBs derived by Jones et al. (1997) using the NAWQC or Tier II SCVs will be used.

Similarly, the dioxins and furans, toxicity equivalent factors (TEFs) have been developed for fish (Van den Berg et al., 1998.) that indicate the relative potency of the individual 2,3,7,8-substituted congeners to the toxicity of 2,3,7,8-tetrachloro-dibenzo-p-dioxin (2,3,7,8-TCDD). These TEFs will be used with the EqP approach to evaluate sediment concentrations of the dioxin and furan congeners.

It is important to note that because of the use of water quality criteria as allowable porewater concentrations, they are likely to be overprotective of benthic organisms. Therefore, they must be used in conjunction with other sediment screening tools such as field observations and bioassays.

NOAA Effects Range-Low and Effects Range-Median Values

If none of the other benchmarks are available for a particular constituents, then the NOAA Effects Range-Low (ER-L) and Effects-Range-Median (ER-M) values will be used. The NOAA ER-L and ER-M values correspond to the tenth and fiftieth percentile of estuarine sediment concentrations reported to be associated with some level of toxic effects (Long et al., 1995). NOAA uses them as concentrations above that adverse effects may begin or are predicted to begin among sensitive life stages and/or species as determined in sublethal tests. It is important to note that because of the limitations used in the development of the NOAA values, the exceedance of a benchmark is not confirmation of an adverse effect, only the indication of the potential for an adverse effect. These limitations, such as the confounding influence of multiple chemicals and the lack of consideration of bioavailability in screening value development, require interpretation of all available site-specific data prior to concluding that an adverse effect does exist.

12.7.3.3 Soil Screening Benchmarks

To determine if the chemical concentrations in surface soils may be toxic to the plant community, the detected concentrations will be compared to Lowest Observed Effects Concentrations (LOECs) as determined in laboratory phytotoxicity studies (Will and Suter, 1995, and Efromyson et al., 1997). The soil screening benchmarks are based on data provided by toxicity studies in the field or more commonly in greenhouse and growth chamber settings. These studies evaluated the effects of chemicals on various trees, wildflowers, grasses, and vegetable species. The plants were exposed to a variety of concentrations, soil types (with differing physicochemical properties), and exposure periods. Measurement endpoints common to the phytotoxicity tests included growth and yield parameters which represent a greater than 20% adverse effect. Growth and yield measurements are direct estimates of the potential impacts to the plant community.

As with sediments, a hierarchical approach to identifying soil benchmarks for screening potential constituents of concern in soils will be used. The benchmarks in order of their evaluation are as follows:

1. ONRL vegetative benchmarks (Efromyson et al., 1997);
2. USEPA Draft Ecological Soil Screening Levels (USEPA, 2000);
3. Canadian Soil Screening Values (British Columbia Regulation 375/96, 1997)
4. Dutch Soil Intervention Values; and
5. NOAEL/LOAEL identified in literature sources such as Eisler (2000a, 2000b, and 2000c).

12.7.4 Toxicity Tests

The ERA will use laboratory surface water and sediment bioassays conducted on samples collected from the Mississippi River to evaluate the toxicity of these two matrices to fish and macroinvertebrates. The results of the toxicity tests will be correlated with other lines of evidence in the respective matrix to develop exposure-response relationships. The exposure-response relationship will be developed using gradients across all sampling locations.

12.7.5 Wildlife Screening Benchmarks

Potential ecological impacts to wildlife ROIs will be assessed by comparing the exposures to ROIs (calculated using a simplified food chain models) to wildlife benchmarks identified from the literature. The wildlife benchmarks are toxicity reference values (TRVs) that are derived from available toxicological data.

Two TRVs will be derived for each chemical and wildlife ROI. One will be based on the no-observed-adverse-effect-level (NOAEL) and the other will be based on the lowest-observed-adverse-effect-level (LOAEL). The NOAEL corresponds to the highest dose at which no adverse effects on growth, reproduction, or survival have been observed. The LOAEL corresponds to the lowest dose at which adverse effects on growth, reproduction or survival have been observed. The ERA will indicate whether the NOAEL and LOAELs are bounded or unbounded. Bounded values are those data sets where a NOAELs and LOAELs has been determined for a certain chemical. An unbounded data set is an NOAEL for which there are no LOAELs, thereby adding uncertainty by not having an identified point at which a chemical may first illicit an effect. Unbounded TRVs will be identified as a source of uncertainty. TRVs for wildlife ROIs (both aquatic and terrestrial) will be reported in units of COPEC exposure per unit bodyweight per day (mg COPEC/kg bw-day) for a specified effect to the receptor. For the ERA, TRVs for wildlife ROIs will be derived from laboratory study results by generally following the methodology of Sample et al. (1996). The following literature sources will be used in the selection of the TRVs:

- U.S. Fish and Wildlife Service biological reports prepared by R. Eisler;
- Toxicological studies cited in Sample et al. (1996);
- U.S. Army Corps of Engineers Waterways Experiment Station online database;
- Ecotox – Ecological Modelling and Ecotoxicology database by L.A. Jorgensen, S.E. Jorgensen, and S.N. Nielsen, Elsevier Publishers, 2000;
- Computer online databases, such as Toxline, Biosis, Wildlife Fisheries Review, Pollution Abstracts, and Environmental Abstracts;

- USEPA Ecotox database; and
- Other readily available literature.

When reviewing the toxicological literature and selecting the most appropriate study for TRV development, several factors will be considered, including:

- Taxonomic relationship between the test animal and the indicator species;
- Use of laboratory animals or domesticated species;
- Ecological relevance of the study endpoints; studies with toxicity endpoints, such as reproduction, growth, behavior and developmental endpoints will be targeted. Sensitive endpoints such as reproductive or developmental toxicity will be preferentially selected because they are closely related to the selected assessment endpoints;
- Toxicological studies in which the chemical was administered through the diet of the test species will be preferred over studies using other oral dosing methods, such as gavage; and
- Long-term studies representing chronic exposure will be preferentially selected.

Species specific toxicity studies may not be available for all COPECs. For mammalian species, smaller animals have higher metabolic rates and are usually more resistant to toxic chemicals because of more rapid rates of detoxification. It has been shown that metabolism is proportional to body surface area that, for lack of direct measurements, can be expressed in terms of body weight (bw) raised to the 3/4 power ($bw^{3/4}$) (Travis and White, 1988; Travis et al., 1990; and USEPA, 1992b). If the dose (d) itself has been calculated in terms of unit body weight (i.e., mg/kg), then the dose per unit body surface area (D) equates to:

$$D = \frac{d \times bw}{bw^{3/4}} = d \times bw^{1/4}$$

The assumption is that the effective dose per body surface area for species "a" and "b" would be equivalent. Therefore, knowing the body weights of two species and the dose (d_b) producing a given effect in species "b," the dose (d_a) producing the same effect in species "a" can be determined. Using this approach, if a NOAEL is available for a test species ($NOAEL_t$), the equivalent NOAEL for a wildlife species ($NOAEL_w$) will be calculated using the adjustment factor for differences in body size:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{1/4}$$

This methodology is equivalent to that the USEPA uses in their carcinogenicity assessments and Reportable Quantity documents for adjusting from animal data to an equivalent human dose.

For the dioxins and furans, toxicity equivalent factors (TEFs) have been developed for mammalian wildlife receptors (Van den Berg et al., 1998.) that indicate the relative potency of the individual 2,3,7,8-substituted congeners to the toxicity of 2,3,7,8-tetrachloro-dibenzo-p-dioxin (2,3,7,8-TCDD). These TEFs will be used to develop congener-specific TRVs to evaluate the potential effects to mammalian wildlife receptors of the 2,3,7,8-substituted dioxin and furan congeners.

For avian receptors, however, there are reports in the literature (e.g., studies cited in Sample et al.. 1996) that state that the above relationship for the majority of chemicals reduces to a factor of one (exponent = 0). That is, there does not seem to be a scaling related to body weight or surface area for birds.

For the dioxins and furans, toxicity equivalent factors (TEFs) have been developed for avian wildlife receptors (Van den Berg et al., 1998.) that indicate the relative potency of the individual 2,3,7,8-substituted congeners to the toxicity of 2,3,7,8-tetrachloro-dibenzo-p-dioxin (2,3,7,8-TCDD). These TEFs will be used to develop congener-specific TRVs to evaluate the potential effects to avian wildlife receptors of the 2,3,7,8-substituted dioxin and furan congeners.

In cases where a NOAEL for a specific chemical is not available, but a LOAEL has been determined experimentally, or where the NOAEL is from a subchronic study, the chronic NOAEL will be estimated. USEPA (1993) suggests the use of uncertainty factors of 1 to 10 for subchronic NOAEL to chronic NOAEL and LOAEL to NOAEL estimation. Based on the available literature, uncertainty factors will be derived for extrapolating from study results to wildlife chronic NOAEL and LOAEL benchmarks.

If toxicological animal studies are not available for a particular COPEC, then quantitative structure activity relationships will be considered and a surrogate chemical will be selected when possible. If the COPEC can not be assessed quantitatively, then the risks associated with the constituent will be qualitatively discussed.

TRVs for each constituent and each ROI will be submitted to USEPA for review and approval prior to preparation of the ERA. In order to conservatively assess the potential for ecological risks from the presence of PCBs in the terrestrial environment, mink (*Mustela vison*) TRVs for PCBs will be used as a surrogate TRV for the short-tailed shrew.

12.7.6 Wildlife Exposure Models

For the wildlife ROIs, a generalized exposure model will be utilized to estimate exposure to the selected wildlife species. Exposure to each COPEC will be estimated by calculating an average daily dose (ADD) using (1) exposure media-specific concentrations, (2) estimated or measured exposure-point concentrations for prey, and (3) receptor-specific exposure parameters. The ADD represents the amount of a chemical that an individual member of a receptor population would ingest if the individual foraged at least a portion of the time within the area used to develop exposure-point concentrations.

Exposure point concentrations for wildlife receptors are best expressed as average COPEC concentrations in prey, surface water, and sediment or soil within the receptors' foraging area. Air is not considered a pathway of concern for this ERA. ADDs for ROIs will be developed based on procedures outlined in Sample et al. (1996) and USEPA (1993). For this ERA, it is assumed that food items are obtained from a particular area and environmental medium as a function of an Area Use Factor (AUF) and a Seasonal Use Factor (SUF). The AUF accounts for relative foraging times based on the ratio of the amount of habitat containing COPECs from a particular on-Site location to the species-specific foraging area. The SUF accounts for the portion of the year that an ROI may be exposed to Site-related COPECs. The total ADD is the sum of ADDs for each of the pathways (i.e., food, surface water, and sediment and/or soil), adjusted for the seasonal duration of exposure and normalized to body weight.

The surface water input for both the red fox and the mink will use water from both the Mississippi River and from the ponds in Site Q, if water is present and samples are collected. If pond water is included, the water input value will include data from the two water sources presented on an area weighted basis, using the length of shoreline as the basis for the weighing. The formula for calculating the ADD is as follows:

$$ADD = (Dose_{food} + (Dose_{sediment} \text{ or } Dose_{soil}) + Dose_{water}) \times SUF$$

where:

ADD	=	Average daily dose of COPEC (mg/kg BW/day);
$Dose_{food}$	=	Dose of COPEC in food (mg/kg BW/day);
$Dose_{sediment}$	=	Dose of COPEC in sediment (mg/kg BW/day), aquatic ROIs;
$Dose_{soil}$	=	Dose of COPEC in soil (mg/kg BW/day), terrestrial ROIs;
$Dose_{water}$	=	Dose of COPEC in water (mg/kg BW/day);
SUF	=	Seasonal Use Factor (unitless); and

The individual terms of the equation are calculated as:

$$Dose_x = IR_x \times C_x \times AUF$$

where:

$Dose_x$	=	Dose of the particular medium in mg/kg BW/day or L/kg BW/day;
IR_x	=	Ingestion rate of medium in mg/kg BW/day (wet weight) or L/kg BW/day;
C_x	=	Concentration of COPEC in particular medium in mg/kg or mg/L; (The average concentration of a COPEC from an area of interest will be used as the exposure point concentration, this value will be

replaced with more realistic assumptions or site-specific prey data as appropriate); and

AUF = Area Use Factor (unitless).

The development of exposure estimates will be made for each of the wildlife ROIs, except the red fox, on a site by site basis, using data from each of the five Sites on an individual basis. Data from each sample location will be included in the exposure estimation for that particular site. For the red fox, the five Sites will be considered in total in the determination of exposure estimates. The results of surface soil, surface water, plant, and earthworm chemical analyses will be examined using both the mean and 95th Upper Confidence Limit (UCL) for the determination of exposure. Noting that the five areas are not contiguous, an area weighted value will be applied to the Sites in relationship to the entire area east of the Mississippi River, south of the MacArthur bridge railroad tracks, west of Illinois State Highway 3, and north of Cargill Road.

12.7.6.1 Exposure Model Input Parameters

Receptor-specific exposure parameters from the scientific literature will be used estimate the ADD for each ROI. Receptor-specific exposure parameters include body weight, food ingestion rate, water ingestion rate, and incidental sediment or soil ingestion rate. An additional exposure parameter, foraging habitat and range will be used to determine the size and type of areas for assessment. The species-specific life history parameters used to calculate exposure for various ROIs are listed below. All exposure parameters are from USEPA (1993) and/or Sample and Suter (1996). The primary citation is indicated adjacent to the parameter.

To estimate the potential COPEC exposure concentrations for the mink foraging either along the Mississippi River or at the edges of the ponds, the following assumptions will be made:

- Body weight = 550 g (using female as sensitive endpoint; Mitchell, 1961);
- Food ingestion rate = 0.137 kilograms/ day (Bleavins and Aulerich, 1981);
- Sediment consumption = negligible (Sample and Suter, 1994);
- Water consumption = 0.099 L/d (Sample and Suter, 1994);

- Diet consists of 100% fish (actual diet composition is dominated by mammals (46%), then fish (15%), amphibians (13%), and birds (8%); however for purposes o f this ERA, the diet will be assumed to be 100% fish, Sample and Suter, 1994);
- Home range = 770 hectare (range size and shape depends on habitat, being usually linear along streams and circular in marshes, USEPA, 1993);
- SUF = 100%;
- AUF = 100% (conservatively assumes that foraging range equal to the length of shoreline abutting the Sites; USEPA, 1993); and
- Habitat = Mink are found associated with aquatic habitats of all kinds, including waterways such as rivers, streams, lakes, ditches, swamps, marshes and backwater areas (USEPA, 1993).

To estimate COPEC exposure concentrations potentially experienced by the osprey foraging along the Mississippi River, the following assumptions will be made:

- Body weight = 1,568 g (using female as sensitive endpoint; Brown and Amadon, 1968);
- Food ingestion rate = 0.21 g/g of BW/d (Poole, 1983);
- Sediment consumption = negligible (USEPA, 1993);
- Water consumption = 0.051 g/g of BW/d (USEPA, 1993);
- Diet consists of 100% fish (USEPA, 1993);
- Foraging radius = 1.7 km (Dunstan, 1973);
- SUF = 100%
- AUF = 25% (assume foraging range equal to four times the size of the Sites; Dunstan, 1973); and
- Habitat = USEPA (1993) reports that the majority of osprey populations in the United States are associated with marine environments, but large inland rivers, lakes and reservoirs may support this species. Habitat useage is dependent upon the presence of good nesting sites (e.g., tops of isolated and often dead trees and man-made structures) in proximity to open, shallow water with a plentiful supply of fish. USEPA (1993) further reports that osprey are almost exclusively piscivorous and that osprey will most successfully feed on slow-moving fish that eat benthic organisms in shallow waters and fish that remain near the water's surface.

To estimate COPEC exposure concentrations potentially experienced by the short-tail shrew foraging within the Sites, the following assumptions will be made:

- Body weight = 0.015 kg (Schlessinger and Potter, 1974);
- Food ingestion rate = 0.009 kg/d; Barrett and Stuek, 1976);
- Soil consumption = 0.00117 kg/d (Talmage and Walton, 1993);
- Water consumption = 0.033 L/d (Chew, 1951);
- Diet consists of 31.4% earthworms (the remainder is reported as being insects and plants, Whitaker and Ferraro, 1963), for purposes of this ERA, the diet of the short-tailed shrew will be considered to consist of 67% of earthworms and 33% of other terrestrial invertebrates, chemical residue data from the earthworms will be developed using bioassay/bioaccumulation tests using soil samples from the Sites, terrestrial invertebrate residue data will be developed through the laboratory analysis of composite samples from the Sites);
- Home range = 0.39 ha (Buckner, 1966);
- SUF = 100%;
- AUF = 100% (This is a conservative assessment that assumes that the shrew will spend it entire life within an area containing elevated COPEC concentrations and that the limited number of soil samples that will be collected are representative of soil conditions throughout a given area); and
- Habitat = USEPA (1993) notes that short-tailed shrews inhabit a variety of habitats and are common in areas with abundant vegetative cover, though they require cool, moist habitats because of their high metabolic rates and water-loss rates.

To estimate COEPC exposure concentrations potentially experienced by the prairie vole foraging within the Sites, the following assumptions will be made:

- Body weight = 41.6 g (Abramasky and Tracy, 1980);
- Food ingestion rate = 0.10 g/g of BW/d (Dice, 1922);
- Soil consumption = 0.00012 (Beyer et al., 1994);
- Water consumption = 0.21 g/g of BW/d (Dice, 1922);
- Diet consists of 100% plant material (USEPA, 1993);

- Home range = 0.098 ha (Jike et al., 1988);
- SUF = 100%;
- AUF = 100% (This is a conservative assessment that assumes that the prairie vole will spend it entire life within an area containing elevated COPEC concentrations and that the limited number of soil samples that will be collected are representative of soil conditions throughout a given area); and
- Habitat = USEPA (1993) reports that the prairie vole inhabits a wide variety of prairie plant communities and moisture regimes, including riparian, short-grass and tall-grass prairie communities. They generally prefer areas of dense vegetation and their presence in a habitat depends upon the presence of suitable cover for their runways.

To estimate COPEC exposure concentrations potentially experienced by the red fox foraging within the Sites, the following assumptions will be made:

- Body weight = 4.13 kg (using female as sensitive endpoint; Storm et al., 1976);
- Food ingestion rate = 0.14 g/g of BW/d; Sargeant, 1978);
- Soil consumption = 0.0126 kg/d (Sample and Suter, 1984);
- Surface water consumption = 0.38 L/d (Sample and Suter, 1994);
- Diet consists of 68.8% mammals and 10.4% plants (the remainder of the diet is assumed to be uncontaminated insects, birds and miscellaneous items, Sample and Suter, 1994);
- Territory size = 96 ha (using female as sensitive endpoint; Ables, 1974);
- SUF = 100%;
- AUF = 100% (This is a conservative assessment that assumes that the red fox will spend it entire life within an area containing elevated COPEC concentrations and that the limited number of soil samples that will be collected are representative of soil conditions throughout a given area); and
- Habitat = Red foxes are the most widely distributed carnivore in the world and will inhabit areas ranging from arctic habitats to temperate deserts (USEPA, 1993). They prefer broken and diverse upland habitats, such as occur in most agricultural areas.

12.7.6.2 Exposure Model Modifications for the Red Fox

As described above, the generalize exposure model will be utilized to estimate exposure values for each potential COPEC to the red fox based on ingestion of food and water from the Sites. The food parameter described in Section 12.7.6.1 includes both a plant and animal component. While direct measurements of the concentrations of COPECs in plant material will be made for each of the Sites, concentrations in animal prey will not be measured, but will have to be estimated. The estimation of the concentration of COPECs in prey species will be made based on the guidance outlined in USEPA (1999b). A compound-specific bioconcentration factor (BCF) consists of biotransfer factors (Ba) and ingestion rates. A biotransfer factor is the ratio of the compound concentration in fresh (wet) weight animal tissue to the daily intake of the compound by the animal through the ingestion of food items and media (soil, sediment, and/or surface water). The following formula is used to derive the BCFs for the ingestion of prey by the red fox:

$$BCF_{F-A} = Ba_A \times IR_F$$

where:

BCF_{F-A}	=	Bioconcentration factor for food item (prey to red fox) in mg of COPEC/kg FW tissue / mg COPEC/kg FW food item);
Ba_A	=	COPEC-specific biotransfer factor applicable for the animal (day/kg FW tissue) as presented in Appendix D to USEPA (1999b) (As a Ba_A is not presented specifically for the prairie vole, the BA_A for the deer mouse (*Peromyscus maniculatus*) will be used in the development of the BCF_{F-A} values); and
IR_F	=	Red fox ingestion rate (kg FW/day)

The red fox exposure calculations will be based on the following pathways:

1. Soil to plant to vole to red fox;
2. Soil to earthworm to short-tailed shrew to red fox;
3. Soil to plant to red fox; and

4. Soil to red fox through incidental ingestion.

To fill all the terms of the exposure model calculation, direct measurements of soil concentrations of COPECs for incidental ingestion will be used, as will direct measurements of COPECs in plants. Using the measurement of COPECs in plants, the BCF_{F-A} will be applied to the ADD calculated for the voles for each COPEC, as described in USEPA (1999b). That resulting value will then be included in the exposure model calculations as the concentration of each COPEC in animal prey for the red fox.

12.8 Risk Characterization

Risk characterization combines Site-related exposures and the potential for ecotoxicological effects to estimate the likelihood of ecological risks. The risk characterization is conducted for each line of evidence and then a weight-of-evidence approach is used to evaluate effects for each assessment endpoint. For the ideal risk assessment, there are three lines of evidence; literature-derived single chemical toxicity data which indicates the potential effects of the COPEC concentrations measured in site media, biological surveys of the potentially affected system which indicate the actual state of the potentially affected environment, and toxicity tests with ambient media which indicate the potential effects of COPEC concentrations measured in site media.

Procedurally, the risk characterization is performed for each assessment endpoint by (1) screening all measured COPECs against toxicological benchmarks (where possible, exposure-response gradients will be developed to help ascertain a more precise understanding of the potential for impacts to receptors); (2) estimating the potential effects of the COPECs identified at the Site; (3) estimating the effects of ambient media based on the media toxicity test results (if conducted); (4) logically integrating the lines of evidence to characterize risks to the endpoint, and (5) listing and discussing the uncertainties in the assessment.

The screening of measured COPECs or of estimated biological body burdens is conducted by the mathematical comparison of a constituent concentration to an ecotoxicological benchmark. The resulting value is identified as a Hazard Quotient (HQ). HQs are calculated using the following equation:

$$HQ = \frac{\text{Media-Specific Constituent Concentration or Total Estimate of Exposure}}{\text{Ecotoxicological Benchmark (in comparable units)}}$$

If the HQ is less than, or equal to, one, then it is concluded that the potential for impacts to ecological receptors is absent or minimal. If the HQ is greater than one, then it is concluded that a *potential* for impacts to ecological receptors exists. Some authors suggest that HQs not be used as a definitive statement of the presence or absence of an ecological risk. Instead, they argue that the HQ serves as an indicator of order-of-magnitude possibilities for impacts to a receptor. Therefore, they suggest that HQs between 1 and 10 indicate only a slight possibility of an ecological risk, while an HQ in excess of 10 can indicate more a probable chance that an ecological impact will occur.

It is again cautioned that an HQ greater than one is not confirmation that an ecological risk is occurring, only indication of the potential for a risk and the need to examine other lines of evidence for confirmation. The magnitude of the exceedance over one may be related to the magnitude of the potential for impact. The level of conservatism in the benchmark, however, must be taken into consideration as a comparison of chemical data to strongly conservative benchmarks may be overprotective of the resource and exaggerate the potential for an ecological risk. As part of the comparison process, the spatial distribution of exceedances will be examined. Exceedances at a large number of sampling points will indicate the potential for more widespread risks as opposed to the exceedance at a single point which would suggest localized impacts. Calculations of HQs will take into account spatial and temporal factors.

As the exceedance of a HQ indicates the potential for an ecological impact to an individual receptor at a specific location, it is the focus of this ERA to determine the potential for ecological risks to the population. As stated in OSWER Directive 9285.7-28 P (USEPA, 1999b), "Superfund remedial actions generally should not be designed to protect organisms on an individual basis (the exception being designated protected status resources, such as listed or candidate threatened and endangered species or treaty-protected species that could be exposed to site releases), but to protect local populations and communities of biota". To asses that potential, an estimation will be made of the population wildlife receptors at each of the Sites, based on habitat availability, home range of the organisms and size of each of the Sites.

12.8.1 Risk Characterization of the Fish Community

The characterization of risks to the fish community endpoint will be determined through the evaluation of multiple lines of evidence. These lines of evidence will be correlated in an evaluation that will potentially demonstrate relationships between the magnitude of exposure and the magnitude of effects. The lines of evidence to be used in the development of the exposure-response relationship are described below.

12.8.1.1 Surface Water Toxicity Testing

The first and strongest line of evidence is toxicity testing conducted on surface water samples collected from Site-related areas. Surface water toxicity tests will demonstrate the exposure-response relationship between the frequency of significant toxic responses and the magnitude of COPEC concentrations measured in the media. The surface water toxicity test results will be examined in relationship to the detected chemical concentrations in surface water to ascertain whether a gradient of toxicity test results can be developed in correlation to gradients in surface water concentrations. As such an exposure-response gradient should be summarized in terms of spatial distribution (Suter et al., 2000), toxicity tests will be examined longitudinally across the sampling stations (across the river) and latitudinally the length of the river. Test responses used in the surface water assessment include the survival of an invertebrate (*Ceriodaphnia dubia*) and a minnow (*Pimephales promelas*).

12.8.1.2 Analysis of Surface Water Chemical Data

The next line of evidence (both in number and in relative weight) consists of the comparison of COPEC concentrations identified in surface water samples with ecotoxicological benchmarks identified in Section 12.7.3. Concentrations of COPECs detected in surface water will be compared to the benchmarks using the HQ process described in Section 12.8.

12.8.1.3 Analysis of Fish Body Burden Data

Another line of evidence to be used in the evaluation of the fish community will be the fish tissue body burden data that is developed for the Study Area. Tissue data from Study Area fish will be compared to tissue residue data from the literature (e.g., Jarvinen and Ankely, 1999) and to NOAEL and LOAEL benchmarks that indicate potential ecological concerns. The fish tissue data will be examined using the HQ process and the resulting values placed in a exposure-response gradient in correlation with the surface water chemical and bioassay data. Concentrations of COPECs identified in upstream fish tissue will be compared to tissue from Site-related fish in an attempt to apportion Site-related COPEC body burdens.

12.8.1.4 Analysis of Sediment Toxicity, Biological and Chemical Data

Benthic invertebrates found within Site-related sediments will be evaluated as a prey base for the fish community. Should impacts to the benthic community be detected, it is assumed that this would potentially impact the ability of the fish community to utilize this resource for food. The extent of the impacted area will be compared to the foraging ranges of fish species to evaluate potential effects of reduced prey availability. As part of the development of this line of evidence, sediment bioassays will be assessed to determine the direct toxicity of sediments to benthic invertebrates.

Toxicity testing will be conducted on sediment samples collected in accordance with the QAPP and FSP. The sediment toxicity test results will be examined in relationship to the detected chemical concentrations in sediment to ascertain whether a gradient of toxicity test results can be developed in correlation to perceived gradients in sediment concentrations. As such an exposure-response gradient should be summarized in terms of spatial distribution (Suter et al., 2000), toxicity tests will be examined longitudinally across the sampling stations (across the river) and latitudinally the length of the river. Test responses used in the sediment assessment will include the survival of two invertebrates, *Hyalella azteca* and *Chironomus tentans*.

Another line of evidence for evaluating the ability of the benthic community to serve as a prey base for the fish community will be biological surveys of the benthic community. Species richness and abundance metrics will be used in evaluating benthic communities in at the various sampling location. Metric values will be placed in context of exposure-response gradients in concert with the chemical and bioassay evaluation. *However*, great care will be taken in this evaluation as there are innumerable functions related to habitat suitability and non-chemical factors that can effect the distribution, abundance and diversity of benthic communities.

Concentrations of COPECs that are identified in Site-related sediments will be evaluated by comparison to sediment benchmarks using the HQ process described in Section 12.8.

12.8.2 Risk Characterization of Amphibians

Concentrations of COPECs in surface water within the ponds will be evaluated for the potential to impact amphibians breeding in the ponds. Concentrations of COPECs identified with the surface water of the ponds will be compared to ecotoxicological benchmarks considered to be protective of amphibians using the HQ process described in Section 12.8.

12.8.3 Risk Characterization of Piscivorous Wildlife

Risks to piscivorous wildlife will be assessed through the evaluation of a single line of evidence for the three species being evaluated. Using the exposure parameters and food-web modeling methodologies described in Section 12.7.6, fish COPEC body burden data will be utilized to determine potential exposures (ADDs) for each species. ADDs will be compared to wildlife TRVs using the HQ methodology outlined in Section 12.8.

As this assessment is based on conservative exposure parameters that may not be Site-specific, exceedance of the TRV will indicate only the potential for an adverse effect and suggest the need for further analysis to confirm that potential. The magnitude of potential exceedances and the spatial distribution of the potential exceedances will be utilized to assess whether the potential for adverse effects may be to a single endpoint organism or to the local population of that species. Both NOAEL-based TRVs and LOAEL-based TRVs will be used in the evaluation process and will be used to estimate a range of exposures that potentially may result in ecological effects to the exposed wildlife. Additional biological survey data such as the identification of an endpoint species during the field surveys or the identification of data such as Audubon Society Christmas bird counts will be compiled to further assist with the evaluation of potential population versus individual ecological risks.

12.8.4 Risk Characterization of Vermivorous Wildlife

Risks to vermivorous wildlife will be assessed through the evaluation of a single line of evidence. Using the exposure parameters and food-chain modeling methodologies described in Section 12.7.7, earthworm COPEC body burden data will be utilized to determine potential exposures (ADDs) for the ROI. ADDs will be compared to wildlife TRVs using the HQ methodology outlined in Section 12.8.

As this assessment is based on conservative exposure parameters that may not be site specific, exceedance of the TRV will indicate the potential for an adverse effect and suggest the need for further analysis to confirm that potential. Both NOAEL-based TRVs and LOAEL-based TRVs will be used in the evaluation process and will be used to estimate a range of exposures that potentially may result in ecological effects to the exposed wildlife.

12.8.5 Risk Characterization of Herbivorous Wildlife

Risks to herbivorous wildlife will be assessed through the evaluation of two lines of evidence. The more significant line is the analysis of potential food chain impacts to the population of herbivorous mammals in the Sites. Using the exposure parameters and food-chain modeling methodologies described in Section 12.7.6, plant tissue COPEC data will be utilized to determine hypothetical exposures (ADDs) for the ROI. ADDs will be compared to wildlife TRVs using the HQ methodology outlined in Section 12.8.

The second line of evidence will be the evaluation of the plant community to serve as both habitat and a food source to herbivorous wildlife. The evaluation of potential impacts to plants will be completed through the screening of surface soil data against soil benchmarks.

12.8.6 Risk Characterization of Carnivorous Wildlife

Risks to carnivorous wildlife will be assessed through the evaluation of a single line of evidence. Using the exposure parameters and food-chain methodologies described in Section 12.7.6, ADDs for the red fox will be calculated. ADDs will be compared to wildlife TRVs using the HQ methodology outlined in Section 12.8.

This assessment is based on conservative exposure parameters and hypothetical body burdens in prey species. As such it is considered to be a screening-level assessment. The exceedance of an HQ is only the indication of the potential for an adverse effect, and not the confirmation of one. Both NOAEL-based TRVs and LOAEL-based TRVs will be used in the evaluation process and will be used to estimate a range of exposures that potentially may result in ecological effects to the exposed wildlife. Additional analyses would be necessary to further refine the estimates of ecological risk, should they be identified in the development of this ERA.

12.8.7 Uncertainty Analysis

Following the characterization of risks, sources of uncertainty and variability within the ERA will be identified. The impact associated with these uncertainties will be qualitatively addressed. A qualitative sensitivity analyses will be conducted for the more important exposure parameters that are used in the wildlife exposure models and for the TRVs that are used to estimate risks to the representative wildlife species.

12.9 Ecological Risk Assessment Report

The findings of the ERA will be presented in a Baseline Ecological Risk Assessment (BERA) report.

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Appendix I-B

Representative Photographs of Disposal Areas

SITE P

Photograph 1: Vicinity Sampling Site P-1

Photograph 2: Vicinity Sampling Site P-2

Photograph 3: Vicinity Sampling Site P-3

Photograph 4: Vicinity Sampling Site P-4

SITE O

Photograph 5: Sampling Site O-1

Photograph 6: Sampling Site O-2

Photograph 7: Sampling Site O-3

SITE Q

Photograph 8: Sampling Site Q-11

Photograph 14: Sampling Site Q-12

Photograph 13: Sampling Site Q-11

Photograph 9: Sampling Site Q-12

Photograph 10: Sampling Site Q-12

Photograph 11: Sampling Site Q-19